From:	[redacted]
To:	SizewellC
Subject:	RE: The Requirement for an Acoustic Fish Deterrent on the Intakes to Sizewell C NNB - Unique Reference 20025603
Date:	19 May 2021 15:15:49
Attachments:	FGS Presentation of AFD Requirement to Sizewell DCO Public Enquiry - 20 May 2021 - Final.pdf

Good afternoon Siân,

Following on from your last email we have prepared a short presentation to accompany the statement that I will provide tomorrow, and I have attached it to this email. I trust it will provide the Inspectors an easy way to understand the issues surrounding the requirement for an AFD, but obviously I am happy to answer any questions they may have after I have spoken tomorrow.

If there are any other questions then please don't hesitate to come back to me, I am more than happy to help in any way I can.

Best regards,

David

Dr D R Lambert
Managing Director

Fish Guidance Systems

[redacted]

Fish Guidance Systems Ltd, 14 Matrix Park, Talbot Road, Fareham, Hampshire, PO15 5AP
Company Registration Number: 2931612, VAT Number: 631 9251 49

From: SizewellC <sizewellc@planninginspectorate.gov.uk>
Sent: 14 May 2021 08:47
To: David Lambert [redacted]
Cc: SizewellC <sizewellc@planninginspectorate.gov.uk>
Subject: RE: The Requirement for an Acoustic Fish Deterrent on the Intakes to Sizewell C NNB - Unique Reference 20025603

Dear David

If you have a presentation then you can send that to us in advance of the

Open Floor Hearing so that the Examining Authority have sight of it. You can then speak to it at the hearing. It will not be shown during the hearing as this is only for oral representations but it will be published at the next deadline after the hearing.

You will be given 5 minutes to speak at the hearing and the Examining Authority may ask you some questions following that.

Kind regards

Siân Evans
Sizewell C Case Team
National Infrastructure Planning

Helpline: 0303 444 5000
Email: SizewellC@planninginspectorate.gov.uk

Web: https://infrastructure.planninginspectorate.gov.uk (National Infrastructure Planning)
Web: www.gov.uk/government/organisations/planning-inspectorate (The Planning Inspectorate)

Twitter: @PINSgov

From: David Lambert [redacted]
Sent: 12 May 2021 17:21
To: SizewellC <sizewellc@planninginspectorate.gov.uk>
Subject: RE: The Requirement for an Acoustic Fish Deterrent on the Intakes to Sizewell C NNB - Unique Reference 20025603

Good afternoon Siân,

Thank you for getting back to me. I will plan on providing details of the issue at an Open Floor Hearing but would be grateful if you can confirm the best way to present the case please? Do you require just a statement to be read out, or a presentation, or something else? And can you confirm if there will be any questions from the inspectors, or it is simply a presentation of our points?

Lastly, can you confirm if a presentation will be helpful, how long can we expect to have to speak.

Many thanks,

David

Dr D R Lambert

Managing Director

Fish Guidance Systems

T[redacted]

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Company Registration Number: 2931612, VAT Number: 631 9251 49

From: SizewellC <sizewellc@planninginspectorate.gov.uk>
Sent: 12 May 2021 16:44
To: David Lambert [redacted]>
Cc: SizewellC <sizewellc@planninginspectorate.gov.uk>
Subject: RE: The Requirement for an Acoustic Fish Deterrent on the Intakes to Sizewell C NNB - Unique Reference 20025603

Dear Mr Lambert

Thank you for your email. It would be helpful if the Panel could hear you on this matter at an Open Floor Hearing. However you may also wish to attend the relevant ecology Issue Specific Hearing, which will be dealing with many aspects of ecology. Details of the Issue Specific Hearings will be published in due course.

Kind regards

Siân Evans
Sizewell C Case Team
National Infrastructure Planning

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Web: www.gov.uk/government/organisations/planning-inspectorate (The Planning Inspectorate)

Twitter: @PINSgov

From: David Lambert <[redacted]
Sent: 12 May 2021 11:15
To: SizewellC <sizewellc@planninginspectorate.gov.uk>
Subject: The Requirement for an Acoustic Fish Deterrent on the Intakes to Sizewell C NNB - Unique Reference 20025603

Good morning,

As you are aware, I spoke on behalf of Fish Guidance Systems Ltd (FGS) at the preliminary meeting to raise the issue regarding the absence of an Acoustic Fish Deterrent (AFD) as part of the proposed mitigation measures on the intake of the new station. FGS also submitted a document outlining why an AFD should be installed, highlighting that the AFD forms part of screening measures that independent experts have identified as best practice, as well as the potential safety issues associated with fish inundations and also the basic assumption that we should protect the fish that EDF/Cefas also agree will be killed if an AFD is not installed.

Our suggestion was that the requirement should form part of an Issue Specific Hearing (ISH), but we note that the Rule 8 Letter and Annexes do not identify this subject as being covered by an ISH.

I would be grateful if you can confirm whether the Planning Inspectorate intends to consider the requirement for an AFD as part of the review of the cooling water system?

I have registered to speak on behalf of FGS at the Open Floor Hearings next week, but it would seem more appropriate that we are included in the discussion about the cooling water system. If that is the case I assume there isn't a need for me to speak at the Open Floor Hearings, so would be grateful if you can advise on the best way forward.

Many thanks,

Yours faithfully,

David Lambert

Dr D R Lambert
Managing Director

Fish Guidance Systems

[redacted]

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Company Registration Number: 2931612, VAT Number: 631 9251 49

Please take a moment to review the **Planning Inspectorate's Privacy Notice** which can be accessed by clicking this link.

The Case for an Acoustic Fish Deterrent (AFD) at Sizewell C (SZC)

Dr David Lambert - Managing Director
And
Dr Andy Turnpenny - Fisheries Specialist

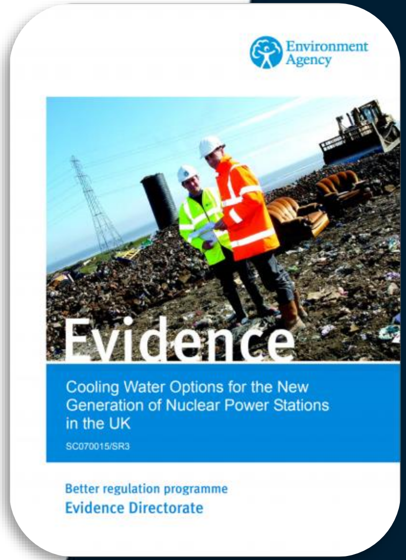
Why Install an Acoustic Fish Deterrent (AFD)?

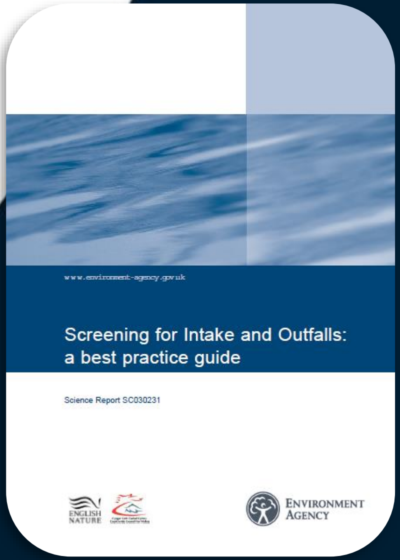
- AFD systems are a **fundamental part of Environment Agency Best Practice** for protecting fish at nuclear seawater cooling intakes
- AFD systems strongly deter fragile fish, such as sprat and herring (88% and 95%)
- Sizewell is known to have large shoals of sprat and herring offshore that will form the bulk of fish drawn into the C station
- Sprat shoals have previously inundated Sizewell intakes, causing loss of cooling water and emergency shutdowns
- EDF/Cefas have confirmed all of the sprat and herring drawn into the intake will die if an AFD is not installed
- Mitigation effects of the proposed Low Velocity Intake Heads are unproven.
- EDF/Cefas have stated that the mitigating effect of the proposed Low Velocity Intake Heads will only be realised if an AFD is installed

What is Best Practice?

Fish Guidance Systems

- The Environment Agency has produced two Best Practice/ Evidence guides for the screening of cooling water intakes
- Four Key Considerations
  - Aim to locate intakes away from sensitive habitat
  - Keep approach velocities low enough for fish to avoid
  - **Ensure fish can detect the intake (e.g. with an AFD)**
  - Provide Fish Recovery & Return (FRR) to maximise survival of fish impinged on CW screens
- **It is fundamental to the Best Practice model that ALL four elements are used together** and is the basis for accepting Direct Seawater Cooling as BAT as per BREF Cooling 2001/EA-SC070015/SR3

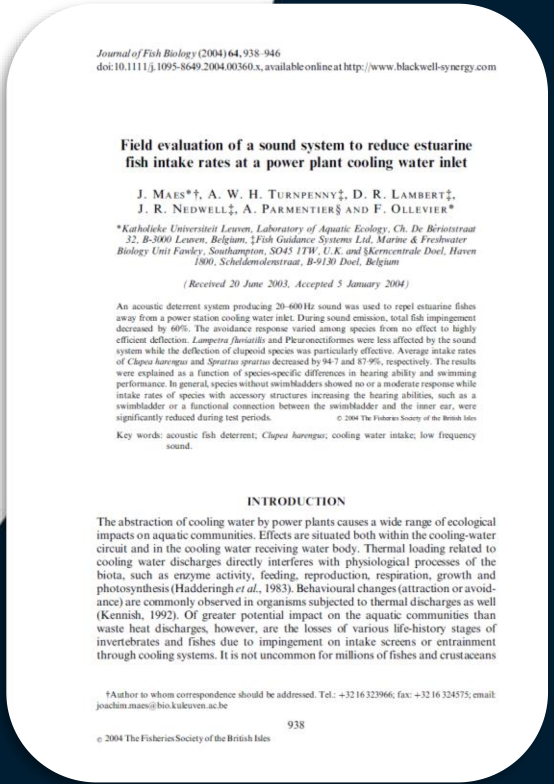




Fish Guidance Systems

How Do AFDs Work?

- AFDs use underwater Sound Projectors to produce a repellent sound field around an intake
- Most fish can hear: Fish such as sprat and herring (clupeids) are particularly sensitive to sound
- Clupeids are also very fragile, any physical contact results in rapid death caused by scale loss and skin damage
- Independent trials have demonstrated that **AFD systems can deter 88% of sprat and 95% of herring**
- AFDs also deter and benefit a wide range of other fish species (**60% overall reduction across 41 fish species**)
- FGS's AFDs incorporate sophisticated monitoring and control systems, enabling real-time feedback on the operation of the system, along with alarms and email / text messaging to advise of any developing faults



Journal of Fish Biology (2004) **64**, 938–946
doi:10.1111/j.1095-8649.2004.00360.x, available online at http://www.blackwell-synergy.com

Field evaluation of a sound system to reduce estuarine fish intake rates at a power plant cooling water inlet

J. Maes*†, A. W. H. Turnpenny‡, D. R. Lambert‡, J. R. Nedwell‡, A. Parmentier§ and F. Ollevier*

**Katholieke Universiteit Leuven, Laboratory of Aquatic Ecology, Ch. De Bèriotstraat 32, B-3000 Leuven, Belgium, ‡Fish Guidance Systems Ltd. Marine & Freshwater Biology Unit Fawley, Southampton, SO45 1TW, U.K. and §Kerncentrale Doel, Haven 1800, Scheldemolenstraat, B-9130 Doel, Belgium*

(Received 20 June 2003, Accepted 5 January 2004)

An acoustic deterrent system producing 20–600 Hz sound was used to repel estuarine fishes away from a power station cooling water inlet. During sound emission, total fish impingement decreased by 60%. The avoidance response varied among species from no effect to highly efficient deflection. *Lampetra fluviatilis* and Pleuronectiformes were less affected by the sound system while the deflection of clupeoid species was particularly effective. Average intake rates of *Clupea harengus* and *Sprattus sprattus* decreased by 94·7 and 87·9%, respectively. The results were explained as a function of species-specific differences in hearing ability and swimming performance. In general, species without swimbladders showed no or a moderate response while intake rates of species with accessory structures increasing the hearing abilities, such as a swimbladder or a functional connection between the swimbladder and the inner ear, were significantly reduced during test periods.

Key words: acoustic fish deterrent; *Clupea harengus*; cooling water intake; low frequency sound.

INTRODUCTION

The abstraction of cooling water by power plants causes a wide range of ecological impacts on aquatic communities. Effects are situated both within the cooling-water circuit and in the cooling water receiving water body. Thermal loading related to cooling water discharges directly interferes with physiological processes of the biota, such as enzyme activity, feeding, reproduction, respiration, growth and photosynthesis (Hadderingh *et al.*, 1983). Behavioural changes (attraction or avoidance) are commonly observed in organisms subjected to thermal discharges as well (Kennish, 1992). Of greater potential impact on the aquatic communities than waste heat discharges, however, are the losses of various life-history stages of invertebrates and fishes due to impingement on intake screens or entrainment through cooling systems. It is not uncommon for millions of fishes and crustaceans

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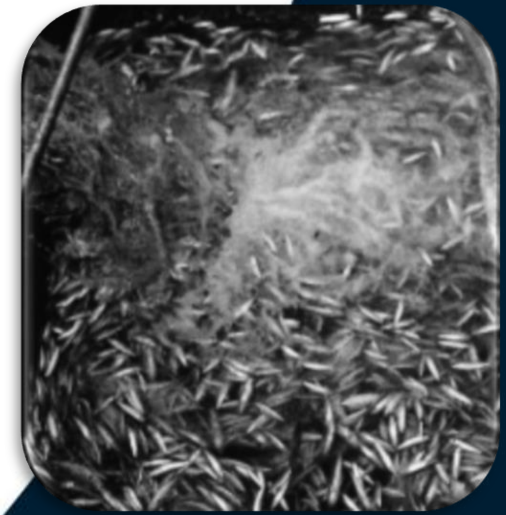
Sizewell and its Fish Communities

- Monitoring of the Sizewell B intake and sampling in the area of Sizewell C intake indicates the fish species drawn into the new intakes will be
  - Sprat (~55%)
  - Herring (~20%)
  - 70 other finfish species
- **~75% of the fish drawn into SZC will be fragile fish**
- Twaite and Allis shad, while not abundant, are protected species and are of high conservation value

Sizewell Sprat Inundations

- The sudden loss of cooling water can lead to emergency shutdowns, and has potentially serious safety implications
- Previous station at Sizewell (Sizewell A) was shut down by a sprat inundation. Such incidents can kill **hundreds of tonnes** of sprat in a single event
- Other power stations on the east coast of England and Scotland have had similar issues in the past
- An EDF nuclear plant in France (Paluel Nuclear Plant) was shut down in January due to a fish inundation
- The risk to the safety of the plant is real, and can be reduced by the installation of an AFD



Sprat drawn into Sizewell A

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The Death of Millions of Fish

- Cefas report TR406 predicts **100% mortality** of all pelagic fish passing through the Fish Recovery and Return (FRR) system. This includes –
    - Sprat
    - Herring
    - Anchovy
    - Twaite shad
    - Allis shad

- Under the proposed Animal Sentience Bill announced in the Queen's speech last week it will formally recognize all vertebrates are sentient beings –and it will support the view that its morally wrong to kill millions of these animals unnecessarily

Low Velocity Intake Heads Unproven

Fish Guidance Systems

- No example of the Low Velocity Intake Heads has been built to date and their effectiveness in reducing fish ingress is unproven. All quoted figures are estimated by Cefas, who are funded by EDF
- Cefas report TR148 (relating to the AFD at Hinkley Point C) states:

"Because of the usual high water turbidity….and the consequent absence of visual clues, ***any mitigating effect of the low-velocity intake is only likely to be realised if it is combined with some form of artificial stimulus*** *(e.g. an acoustic fish deterrent) to induce fish to swim away from the intake structure. Equally however, an acoustic fish deterrent is unlikely to be fully effective on its own if the intake velocity exceeds the swimming capabilities of the fish.* ***For these reasons low-velocity intake and AFD need to be considered as a combined mitigation measure."***

EDF Safety Concerns

- An AFD was originally included in the mitigation measures proposed by EDF, but was removed in Public Consultation 4
- EDF has stated "logistical and safety considerations preclude the use of AFDs at Sizewell C"
- There is 10 years before the AFD will be installed, and these concerns can be overcome in this time
- The AFD system available from FGS meets all the requirements specified by EDF for Hinkley Point C (there is no specification for SZC, but in Public Consultation 3 was assumed by EDF to be similar to the HPC system)
- There are ROVs that carry out similar offshore maintenance work, and if needed can be further developed for the AFD. Divers are not required

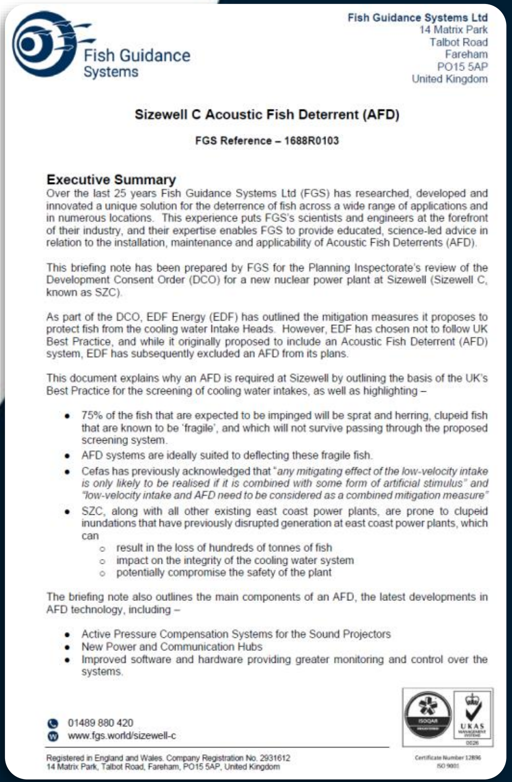
Questions

- Fish Guidance Systems has provided a submission explaining in detail why an AFD should be installed at SZC

- Updates on this project are available at our website –

  fgs.world/sizewell-c

- If anyone has any subsequent questions please do email us at …. info@fgs.world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Sizewell C Acoustic Fish Deterrent (AFD)

FGS Reference – 1688R0103

Executive Summary

Over the last 25 years Fish Guidance Systems Ltd (FGS) has researched, developed and innovated a unique solution for the deterrence of fish across a wide range of applications and in numerous locations. This experience puts FGS's scientists and engineers at the forefront of their industry, and their expertise enables FGS to provide educated, science-led advice in relation to the installation, maintenance and applicability of Acoustic Fish Deterrents (AFD).

This briefing note has been prepared by FGS for the Planning Inspectorate's review of the Development Consent Order (DCO) for a new nuclear power plant at Sizewell (Sizewell C, known as SZC).

As part of the DCO, EDF Energy (EDF) has outlined the mitigation measures it proposes to protect fish from the cooling water Intake Heads. However, EDF has chosen not to follow UK Best Practice, and while it originally proposed to include an Acoustic Fish Deterrent (AFD) system, EDF has subsequently excluded an AFD from its plans.

This document explains why an AFD is required at Sizewell by outlining the basis of the UK's Best Practice for the screening of cooling water intakes, as well as highlighting –

- 75% of the fish that are expected to be impinged will be sprat and herring, clupeid fish that are known to be 'fragile', and which will not survive passing through the proposed screening system.
- AFD systems are ideally suited to deflecting these fragile fish.
- Cefas has previously acknowledged that "*any mitigating effect of the low-velocity intake is only likely to be realised if it is combined with some form of artificial stimulus*" and "*low-velocity intake and AFD need to be considered as a combined mitigation measure*"
- SZC, along with all other existing east coast power plants, are prone to clupeid inundations that have previously disrupted generation at east coast power plants, which can
  - result in the loss of hundreds of tonnes of fish
  - impact on the integrity of the cooling water system
  - potentially compromise the safety of the plant

The briefing note also outlines the main components of an AFD, the latest developments in AFD technology, including –

- Active Pressure Compensation Systems for the Sound Projectors
- New Power and Communication Hubs
- Improved software and hardware providing greater monitoring and control over the systems.

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www.fgs.world/sizewell-c

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Sizewell C Acoustic Fish Deterrent (AFD)
Addendum to Sizewell C Presentation

FGS Reference – 1688R0204

Summary

This addendum is provided in support of the AFD Presentation "FGS Presentation of AFD Requirement to Sizewell DCO Public Enquiry – 20 May 2021 – Final", forwarded to the Planning Inspectorate on 19th May 2021, as well as the original document submitted by FGS to the Inspectorate on 6th April 2021, " Sizewell C Acoustic Fish Deterrent (AFD)", FGS Reference 1688R0103.

It provides the references to support the statements made in the presentation, and the referenced documents are either linked from within or attached to this report.

1. Source of Best Practice Definition for Cooling Water Fish Protection

AFD- Environment Agency guidance

Section 3.4.5 of Environment Agency (2005) fish screening best practice guide describes the use of acoustic fish guidance methods to deter fish from intakes, including principles of AFD, typical system designs (of that period- things have progressed since) and examples of results achieved in reducing fish impingement catch. It goes on the detail implementation of AFD under best practice.

Section 3.4.5.14 of this document describes their particular use at estuarine and coastal seawater cooled power stations, and the complementary function of fish return techniques in putting back to sea the less hearing-sensitive fish species and other biota that are not deterred by sound.

Table 6.2 (p.122) of the guide identifies the combination of acoustic deterrents (AFD), velocity capped intake and fish return (FRR) as best practice for large thermal power plant (including nuclear) with onshore or offshore intakes.

2. Deflection Efficiencies of AFD systems

A Sound Projector Array (SPA) system was installed on the intake of Doel nuclear power plant in 1996, and assessed by Dr Maes of Leuven University.

The paper published in the Journal of Fish Biology in 2004 confirms in the Executive Summary that "*Total fish impingement decreased by 60%"* during the AFD operation, and "*Average intake rates for Clupea harengus* [herring] *and Sprattus sprattus* [sprat] *decreased by 94.7 and 87.9% respectively*".

3. Fish Communities at Sizewell

Reference to the predicted catch rates of sprat and herring as a proportion of the top 24 key species are listed in Table 2 (p.17 of pdf) of EDF/ Cefas Technical Report No. TR406 (EDF Report TR406 Impingement predictions – SZC-SZ0200-XX-000-REP-100070 Revision 6).

Table 2 gives the following predicted annual catches for SZC –

Sprat	7,125,393 individuals per annum
Herring	2,555,783 individuals per annum
Total impingement for all 24 species	12,660,950 individuals per annum

On this basis, sprat account for 56% of the annual catch and herring account for 20%, a total of 76% of these fragile fish. These figures are without any mitigation.

Figures shown in Table 2 with LVSE and FRR mitigation increase the percentage annual catches to 64% for sprat and 23% for herring, resulting in a total of 87% of the annual catch.

In the absence of any proposed AFD, mitigations are ineffective for these species and as noted below in Section 5, they can all be expected to die.

4. Sizewell Sprat Inundations

Notes from Cefas Fisheries Laboratory in 1970 review the seasonal changes in distribution of sprat shoals, and their potential hazard for power station cooling water intakes. On Page 6 (of the pdf) the document reports that "*The Sizewell and Dungeness intakes are both sited in potential danger zones regarding sprat shoals since each is within an area where sprat shoals consistently appear and aggregate each season*"

Page 7 (of the pdf) details the inundation of the Sizewell A intake, and also outlines a similar issue that occurred at Dungeness eight days later.

More recent inundations have occurred at a number of sites along the south coast of England, as outlined in FGS's news article of 17th March 2021.

https://www.fgs.world/news/why-fish-clogging-up-the-intakes-of-nuclear-power-plants-is-very-damaging-for-both-the-environment-and-the-taxpayer/

The latest inundation to hit EDF was at its Paulel nuclear power station on 21st January 2021, as reported in Bloomberg online news –

https://www.bloomberg.com/news/articles/2021-01-21/french-nuclear-plant-halted-after-fish-clog-water-filters

EDF notes in Section 5.7.3 of Report TR406 Impingement predictions – SZC-SZ0200-XX-000-REP-100070 Revision 6 "*The main clogging risk at Sizewell are from winter sprat inundations….*"

EDF proposes to overcome blocking of the screens by using larger 10mm screens, but the installation of an AFD will significantly reduce the number of sprat entering the intake to start with.

5. The Death of Millions of Fish

Section 5.7.2 of EDF Report TR406 Impingement predictions – SZC-SZ0200-XX-000-REP-100070 Revision 6 states that "*survival rates for delicate pelagic species, such herring, sprat and shad (twaite shad Allosa fallax and Allis shad A. alosa) are usually low (<10%, Environment Agency, 2005)*".

Table 2 (p.17 of pdf) of EDF Report TR406 indicates that all of the sprat, herring, anchovy, smelt, twaite shad, sea trout and allis shad are all expected to die if they enter the cooling water system ("FRR mortality" is shown to equal "SZC prediction with LVSE intakes").

Table 2 also indicates that, allowing for the predicted reduction in impingement from the LVSE Intake a mean of 2,729,025 sprat and 978,865 herring will die each year. If the LVSE intakes don't deflect the fish as predicted, then the mean numbers predicted to die increase significantly to 7,125,393 sprat and 2,555,783 herring each year.

Appendix B (p.113 of pdf) of EDF Report TR406 shows that the upper predictions of numbers of fish that may be impinged if the LVSE is not effective, and states that up to 18,206,464 sprat and 23,252,294 herring may die each year.

6. LVSE – unproven to date

Low velocity side-entry intakes (LVSE) are first mentioned in the Environment Agency (2005) best practice guide (Section 3.4.7, p.88). It goes on to say "*Experience at UK sites where velocity caps are present has shown that fish entrainment remains a problem and velocity caps are therefore not in themselves a solution. Other measures, including use of side-entry, of behavioural deterrents and onshore fish return systems need to be considered*".

It is also made clear in this section that the LVSE design and that developers should undertake modelling studies to demonstrate that their intake structures should meet the hydraulic design objectives of the LVSE principle.

Environment Agency (2010) also reasserts: "*As yet, this design has not been built and requires further model testing to develop the detailed design and ensure its suitability*".

EDF has undertaken such modelling studies which indicate a satisfactory design on paper, but no example of an LVSE intake exists worldwide at present. The LVSE design remains therefore unproven with regard to any operational experience or proven benefits to fish.

In addition, in EDF/ BEEMS Technical Report TR148, Section 3.1, it refers to high water turbidity, and the consequential loss of visual clues, and concludes "*For these reasons low-velocity intake and AFD need to be considered as a combined mitigation measure*." The same high-water turbidity can be expected around the coast of the UK and so the same conclusion applies to all other intakes, including Sizewell.

7. EDF Safety Concerns

EDF has raised safety concerns for divers working in high velocity flows in the Bristol Channel for the Hinkley Point C project, and has transferred those concerns over to the Sizewell C project. However, flows are significantly lower at Sizewell than at Hinkley Point.

Section 22.4.4 of the SZC Environmental Statement v.2 CH 22, Marine Ecology and Fisheries

EN010012-001934-SZC_Bk6_ES_V2_Ch22_Marine_Ecology_and_Fisheries.pdf (planninginspectorate.gov.uk) identifies the current regime at Sizewell as follows:
22.4.4 Water movement is dominated by tidal currents that flow south for most of the rising (flood) tide peaking at a velocity of 1.14m/s seaward of Sizewell Bank and flow north for most of the falling (ebb) tide (peak velocity of 1.08m/s). The strong tides and generally shallow bathymetry combine so that the water column is well mixed throughout the year.

The Environmental Statement for HPC (Chapter 19, para 19.6.3: V2 C19 Marine Ecology.pdf (edfenergy.com)) refers to tidal velocities 1 to 5km offshore at Hinkley Point reaching 1.7 m/s.

FGS *personal communications* with a number of ROV engineers has indicated that Remote Operated Vehicles (ROVs) are being developed for use in high flow sites as part of the development of tidal energy projects and the long-term maintenance of offshore windfarms. ROVs are used widely in the oil and gas industry in the North Sea, and there are many articles that outline their benefits. Here are just two –

- https://www.hydro-international.com/content/article/trends-and-new-technology-in-the-rov-industry
- https://www.offshore-technology.com/features/feature1659/#:~:text=The%20North%20Sea%20represents%20about,%2C%20of%20course%2C%20West%20Africa.

In addition, augmented reality for subsea work, including ROV operations, has been called a "game changer" that is becoming more of a reality and very suitable for sites with zero viz.

A number of possible ROVs have been identified, and ROV specialist engineers have offered their services to assist with locating and if necessary developing a suitable ROV to carry out the work without the need for divers. As was noted during the Open Floor Hearings, even if an ROV is not immediately suitable for the project, there is 10 years for EDF engineers to work with suppliers and ROV specialists to develop a suitable deployment system and ROV for the AFD.

8. Documents Attached to this Addendum

Direct links to some articles and reports have been imbedded in this document. Where that has not been possible the following documents have been attached to this document.

- Cefas (1970) – Technical Note on Sprat Shoals at Sizewell
- EDF / BEEMS (2011) - Technical Report TR148 – A synthesis of impingement and entrainment predictions for NNB at Hinkley Point
- EDF / Cefas (2020) - Technical Report TR406 - Sizewell C – Impingement predictions based upon specific cooling water system design – SZC-SZ0200-XX-000-REP-100070 Revision 6
- Environment Agency (2005). Screening for Intake and Outfalls: a best practice guide Science Report SC030231
- Environment Agency (2010). Cooling Water Options for the New Generation of Nuclear Power Stations in the UK SC070015/SR3
- FGS Case Study – Doel Nuclear Power Plant
- FGS Case Study – Pembroke Power Station
- Maes et al (2004). Field Evaluation of a Sound System to reduce estuarine fish intakes at a power plant cooling water inlet. Journal of Fish Biology 64, 938-946

9. Glossary & List of Abbreviations

The following are taken from Environment Agency guidance, along with other additions as required to provide a complete list of acronyms in this document.

AFD: Acoustic fish deterrent: propagation of underwater sounds to deflect fish from water intakes.
Approach velocity: Water velocity just upstream of a screen or water intake.
Band screen: Type of rotating fine filter screen, usually of 3-10 mm mesh, installed upstream of cooling water pumps and condensers to exclude marine detritus. Mesh is formed as 'conveyor' belt that rotates and is continuously backwashed to keep it clean.
BAT: Best Available Technology, as required under European Integrated Pollution Prevention and Control (IPPC) regulations.
BEEMS: British Energy Estuarine & Marine Studies. The BEEMS programme was set up by EFD Energy for Cefas to implement a multi-disciplinary programme of marine studies to support new-build power stations in the UK.
Cefas: Centre for Environment, Fisheries and Aquaculture
CW: Cooling water/circulating water. Latter used mainly with tower cooling.
DCO: Development Consent Order.
Drum screen: Type of rotating fine filter screen, usually of 3-10 mm mesh, installed upstream of cooling water pumps and condensers to exclude marine detritus. Mesh is formed as drum that rotates and is continuously backwashed to keep it clean.
EA: Environment Agency
EDF: EDF Energy
Entrainment: Passage of entrapped organisms that penetrate CW screens (typically zooplankton including ichthyoplankton and phytoplankton), via pumps, heat exchangers and other components of the CW circuit and back to the receiving water.
Entrapment: Inadvertent entry into the CW system of aquatic organisms caused by the ingress of water.
FGS: Fish Guidance Systems Ltd.
FRR: Fish recovery and return: system using band or drum screens modified for safe fish handling, including their return to the source water body.
HPC: Hinkley Point C.
Impingement: Retention of entrapped organisms on CW intake screens employed to prevent debris entering the CW heat exchangers.
Launder: Troughs, channels or pipes used to carry trash that has been backwashed from the fine screens.
LVSE: Low-Velocity Side-Entry, the design of Intake Head being installed at Hinkley Point C and Sizewell C.
m/s: meter per second
NNB: Nuclear New Build Generation Company; NNB is a subsidiary created by EDF Energy to build and then operate Sizewell C
ROV: Remote Operated Vehicle
SPA: Sound Projector Array, a low frequency underwater transducer manufactured by FGS.
SZC: Sizewell C

Dated: 1st June, 2021

2.8.7 SESTON

DUNGENESS
SIZEWELL
IMPINGEMENT
2.8.5

NOTES ON THE SEASONAL CHANGES IN DISTRIBUTION OF SPRAT SHOALS WITH SPECIAL REFERENCE TO THEIR POTENTIAL HAZARD FOR POWER STATION COOLING-WATER INTAKES

The sprat has a widespread distribution around the coasts of the British Isles and shows very pronounced seasonal changes in the local density of its shoals. It is a fairly short lived species (maximum age around 5 years) with 2 and 3 year old fish making up the largest proportion by weight of the population. It can thus show very large changes in population abundance from year to year, depending on the relative success or otherwise of each incoming year class. In addition to true changes in population abundance, apparent changes may occur due to seasonal shifts in the distribution of shoals. The usual pattern of change involves inshore movement during the autumn (October-November) leading to aggregation of the shoals during the winter months in coastal bays and estuaries. The shoal density may then locally increase to a level many times greater than that found at other times of the year. Along the North Sea and Channel coasts the peak aggregation period usually falls between November and March, whilst in the latter month the shoals generally disperse seawards again for spawning. The reasons for this overwintering phase inshore are not clear as very little food is taken over this period, but it may provide a mechanism for aggregating fish prior to spawning. However, not all sprat shoals move inshore at this time, some remain well out to sea during the winter months. The localities along the coast where these shoal concentrations are likely to develop each year are fairly consistent. The major concentration usually build-up within the deeper water channels of estuaries such as the Wash and Thames, whilst off the north-east coast they can appear in the open sea some distance off Tynemouth. On occasion these very dense and extensive patches of fish have appeared close inshore off the Durham and Lincolnshire coasts, although they are less frequently found along the more exposed coasts, the shoals usually remaining fairly well dispersed in such localities, although tending to confine themselves within a few miles from the coast.

Along the Suffolk coast shoals are usually first located between Lowestoft and Southweld in early November and build up to a peak density in December, meanwhile showing a gradual southerly shift towards the Thames Estuary. It is possible that these fish are on passage and this sector of the coast represents a southerly migration route. The local shoal density does not normally reach very high levels and the shoals remain fairly thinly spread within a 5-10 mile wide belt of coastal water. The water is very shallow and turbid in this region and the usual fishing gear used for capturing sprats is the drift net, this gear also being employed along the Kent-Sussex coast.

THE DEPTH OF SPRAT SHOALS AND CHANGES IN VERTICAL DISTRIBUTION

Drift nets provide some evidence on the depth at which the shoals are swimming. On the Suffolk coast they are worked in daylight, or very close to the surface. The nets hang to $2\frac{1}{2}$ - 3 fathoms beneath the surface and the fish generally mesh within the lower half of the nets, mainly 1-2 fathoms beneath the surface. Echo-survey records confirm that in the shallower water (3-5 fathoms) they are very close to the surface, whilst in deeper water (5-10 fathoms) they may be between 2-5 fathoms below.

Along the Kent and Sussex coast the shoals are usually found at a greater depth, the drift-nets worked in this region being provided with "strops" to enable them to be set at varying depths depending on the level at which the fish are swimming. Echo-records from this locality show the shoals commonly located within a depth range 5-9 fathoms beneath the surface in 12-20 fathoms depth of water. Shoals are usually found closer to the surface as one progresses inshore into more turbid water.

However, these observations apply to daylight, and there are also marked diurnal changes in the vertical distribution of fish, although these become less obvious in very shallow and highly turbid water. Towards dusk the shoals will slowly move up towards the surface and eventually disintegrate into scattered individuals which may merge with those from neighbouring shoals to form a diffuse layer in the upper part of the water column during darkness. Towards first light the fish reaggregate into discrete shoals near the surface and then gradually descend with increasing light intensity, finally attaining a maximum depth at full light, this probably being dependent on conditions of water turbidity and the level of incident daylight at the time. In full daylight the depth of the shoal layer is usually maintained within fairly small limits, depending on the environmental factors previously mentioned. There is also some degree of vertical segregation of fish by size, in that the shoals of smaller individuals do not normally descend to as deep a level as the larger ones, and this often results in a "double-layered" echo-record. The very smallest sprats (whitebait) are often to be found in the top fathom or so even in full daylight, when the water is very turbid.

There have also been occasions in very dull and overcast weather conditions when little difference has been evident between the day and night echo-records, the fish remaining very dispersed, presumably due to light levels being too low for effective shoal formation.

The normal depth range occupied by a shoal layer will be of particular importance to the siting of a fixed intake. If it can be sited lower than the maximum depth to which the shoal layer might be expected to descend, then the risk of catching any great quantity of these fish becomes reduced to a minimum. As noted earlier, the actual depth at which a shoal layer is located is probably dependent on a number of variables such as size of fish, water turbidity and incident light intensity. The turbidity factor could be influenced by strength and direction of wind, velocity and state of tide, i.e. whether ebb or flood, spring or neap and local condition of run-off. The levels of the intake and shoals may only coincide under certain combinations of circumstances. For instance, if the shoal layer tends to maintain itself within fairly small limits at a fixed depth with reference to the surface, and the intake is sited in a locality showing considerable tidal range, then the danger point might very well fall towards low water, when the two levels come into juxtaposition, assuming that for most of the tidal cycle the shoals are passing above the intake level.

There is no reliable 'rule of thumb' method for estimating the likely maximum depth shoals may descend to in any particular locality, although in water less than 15 fathoms depth which is fairly turbid they are usually recorded in daylight within the upper half of the water column. However, within the range of depths likely to be encountered around the shorelines of British coasts there is unlikely to be an absolute depth barrier to vertical movement since they have been recorded in daylight within the depth range of 40-50 fathoms in deeper, clearer water.

THE SIZE OF SPRAT SHOALS AND ESTIMATES OF THEIR WEIGHT

Evidence from echo-survey records show that discrete shoals can vary in size from very small units measuring some 6 feet in diameter by about 3 feet in vertical thickness, up to much larger units several hundred feet in diameter with a vertical thickness of 30 feet.

Very much larger 'shoal aggregates' are also found, these probably representing shoals in such close proximity to each other that they are no longer distinguishable as independent units within the resolution power of the echo-sounder. These appear as an almost continuous layer of fish which can extend over many miles and form quite discrete patches with fairly clear-cut boundaries. These patches are often very elongate, measuring some 4-5 miles along their major axis and between 1-2 miles in width. On occasion, when shoals are very abundant, these concentrations have been found extending over a distance of 12-15 miles.

These major 'shoal aggregates' are normally found in deeper, more open water off the north east coast of England and within the deeper water channels of the Wash and Thames estuaries. They are less commonly found along more open coasts inshore, although they can occur in such localities in some seasons, either building up here or moving into these areas en masse after forming up elsewhere.

The actual quantity of fish within a shoal of given external dimensions could vary considerably, depending on the packing density of fish within. It is difficult to accurately estimate packing density, direct observation methods would be virtually impossible under turbid water conditions, high resolution acoustic gear may ultimately provide a means, but at present only approximate assessments may be made using reasonable assumptions or results from fishing operations.

Assuming that maintenance of shoal cohesion is dependent on visual contact between individuals, then in turbid water the spacing factor (expressed as fish lengths apart) is not likely to be very great, and for a small fish such as sprat, which averages some 10 cm (4") in length, could fall between 1 and 3 fish length units. If the shape of individual shoals (as distinct from 'shoal aggregates') is taken as an oblate spheroid, then estimates of numbers and hence weight of shoals of given dimensions may be arrived at. The following table provides such estimates for a range of different shoal sizes and fish length spacings. This uses a normal distribution of fish length about a mean of 10 cm with a standard deviation of 1.1 cm. The average weight per fish in such a distribution is 7.5 grms.

ESTIMATES FOR THE WEIGHT OF DIFFERENT SIZED SPRAT SHOALS

(The dimensions refer to the horizontal and vertical diameters of the shoals respectively)

	Packing Density (Fish Lengths Apart)	Total Numbers of Fish	Equivalent kg	Weights cwt
a) 6' x 3'	1	2,915	21.9	0.43
	2	364	2.7	0.05
	3	108	0.8	0.02
b) 12' x 6'	1	23,319	174.9	3.44
	2	2,913	21.8	0.43
	3	864	6.5	0.13
c) 24' x 12'	1	186,553	1,399.1	27.54
	2	23,304	174.8	3.44
	3	6,912	51.8	1.02
d) 48' x 18'	1	1,119,316	8,394.9	165.24
	2	139,824	1,048.7	20.64
	3	41,472	311.0	6.12

The above range of shoal sizes encompasses those commonly found in turbid coastal water. This table shows that the range in weight between the smallest and largest shoals could be quite considerable, results from commercial fishing operations confirming that these estimated values are not unrealistic.

When shoals are densely aggregated it is not uncommon for catch rates of up to and exceeding one ton per minutes' towing time being achieved by small pair-boats towing a mid-water trawl at about 2 knots through the water. In operation this net has a spread of about 36 feet and a gape of 24 feet and the cross sectional area of the aperture is thus some 864 square feet. The average density of fish in the path of the net can thus be estimated from the total numbers of fish caught and volume of water filtered per unit time. This will probably represent a considerable underestimate of the true density of fish within the shoals since it assumes that all fish in the path of the net are caught and the net is continually passing through fish. The gape of the net may also exceed the vertical thickness of the fish layer.

At a towing speed of 2 knots the volume of water filtered in one minute will be about 175,000 cubic feet. One ton of fish (with an average length weight of 7.5 grms) represent about 135,500 individuals, which gives a packing density of 0.8 fish per cubic foot, which is equivalent to an average space between individuals of around 4 fish lengths (using an average of 10 cm). This density is somewhat less than the range used for the estimates of shoal weights, but as noted above is probably considerable underestimate of the actual density within shoals.

The next stage is to consider the likely densities of shoal aggregations. It was noted earlier that at very high levels of abundance shoals can aggregate in such close proximity to each other that they virtually merge to form almost continuous shoal layers which may extend over distances of many miles. Under these circumstances the total weight of fish accumulating within a square nautical mile could reach many thousands of tons.

If for example the largest type of shoal in the above table (48' diameter and 18' thickness) were to form up into a contiguous layer of such shoals (that is touching but not merging) then estimates for the total tonnage of fish within one square nautical mile and over a distance of one nautical mile (single shoal width) work out as follows:-

Packing Density Within Shoal	Total Tons/N. Mile^2	Tons/N. Mile
1 Fish Length	132,500	1,046
2 " "	16,595	131
2 " "	4,940	39

Thus at a speed of 2 knots the linear encounter rate over a shoal width frontage would be at a maximum 35 tons per minute and minimum 1.3 tons per minute. However, shoal densities are normally much less than this, and in open coastal waters such as the Suffolk and Kent/Sussex areas the shoal frequency, in terms of the horizontal proportion of the total echo-record occupied by shoals, is between 20-30% in the zones of maximum shoal density (the above level would represent 100%).

SPEED OF SHOAL MOVEMENT

The sprat being a small fish is not likely to show a very high sustained swimming speed. The size range in winter shoals between about 5-15 cm (2-6"), with a mean size of around 10 cm (4").

The sustained swimming speed of a fish will be about 3 times its own length per second, which gives a speed of 0.5 knot for an average size sprat. For short periods of fish may swim at speeds up to 12-15 times its length per second, which gives an upper limit of 2.0 - 2.5 knots. Shifts in the location of sprat shoals are observable during the winter months, but these shifts are normally fairly slow and the general level of activity is low at this time. It is not always clear when these movements do take place how much is due to the shoals own volition and how much is more or less passive drift brought about by wind and residual currents. However, the autumnal period prior to the winter aggregation period must involve considerable movement of fish with some positive orientation, perhaps assisted by residual drift systems. Nevertheless, it is likely that in regions with strong tidal streams the short-term movement of these shoals can be equated with the direction of velocity of the tidal stream. Evidence for this is provided by the now extinct stownet, formerly widely used to catch these fish. This was virtually an anchored mid-water trawl which was set to fish between slack water periods, the tidal flow carrying fish into the net. Results from echo-surveys have also shown clear tidal displacements of major concentrations, these being readily located and plotted especially when they show fairly clear-cut boundaries.

The incident at Dungeness took place about a week later and was reported as near low water on the afternoon of January 18th, the time of low water being around 18.00 hours (GMT). It seems to have been almost a 'carbon copy' of that at Sizewell and it is of interest to examine the weather factors in the Dungeness area prior to and at the time of this incident to determine whether or not there are any similarities (apart from the 'low water' timing in both cases). The wind had been predominately blowing from a South-South-Westerly direction at 15-20 knots in the period 13-15 January, on the 16th it was WNW, and on the 17th it showed rapid changes in direction and speed until on the 18th it was WSW'ly at 25 knots. It was thus blowing obliquely onshore with probably fairly heavy swell conditions and turbid water, prior to this there had been several days with it blowing more directly onshore, which does correspond in some respects with the condition preceding and during the Sizewell incident.

The Sizewell and Dungeness intakes are both sited in potential danger zones regarding sprat shoals since each is within an area where sprat shoals consistently appear and aggregate each season, although in very variable quantities. Basically the danger level will be dependent on the actual abundance of shoals, which in turn could be influenced by availability changes or the abundance of particular year classes. Very strong year classes can appear at fairly short or much longer intervals, but on average one may be expected to appear each 3-5 years.

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It is thus likely that the main direction of approach of shoals towards a fixed point will be from the 'upstream' direction during the main flow of tide. If the shoals are showing an independent direction and rate of movement this will only become apparent towards and around the slack water period.

ASSESSMENT OF THE CIRCUMSTANCES INVOLVED IN THE INCIDENTS AT SIZEWELL AND DUNGENESS POWER STATION IN JANUARY 1969

The first point of note is that a very abundant year class of sprat (1967) was present as two-year-old fish at this time, these being particularly strongly represented along the east and south-east coasts of England, constituting over 70% of the catch off the Suffolk coast. It first became evident from very heavy drift-net landings in the second week of November 1968 that a very good concentration of shoals had begun to build up along the Suffolk coast. A research vessel survey carried out in mid-December showed the shoals mainly confined within a 10 mile wide belt of coastal water, the main concentration area then extending between Southwold and Aldeburgh, extending over a distance of about 6 nautical miles, the shoal density increasing towards the coast. During the second week of January there were signs of a fairly rapid movement of sprats into the north-east approaches to the Thames Estuary with a build-up of concentrations in the channels along the north side of the estuary. This was also the time when the incident took place at Sizewell power station. It was then reported that a great quantity of sprat had entered the cooling water intake around the time of low water on the morning of January 10th, the fish rapidly clogging the filter screens to such an extent that the combined weight of fish and 'locked up' water was sufficient to break through the screens into the plant. Early the following morning a local coast watcher reported that an extensive layer of dead sprats had been washed ashore along the tide line over a distance of several miles between Dunwich and Thorpeness. The local Inspector of Fisheries later estimated after examining the situation that fish had been washed ashore over a total distance of 6-7 miles and the density amounted to around 1 bushel basketfull (0.5 cwt) every 10-12 yards. Using these estimates it was calculated that at most 30 tons of fish would have been involved to spread over 7 miles of shoreline at this density. The quantity of fish removed from within the power station and dumped on surrounding marshes was not known.

Prior to this incident a fresh to strong South-easterly wind had been blowing for several days, easing the day before for a short while but then increasing again towards the end of the day, until about the time of the occurrence it was fresh to strong (20-25 knots) East-South-Easterly with very dull overcast conditions. Wind from this quarter in this region tends to generate heavy swell conditions and coastal erosion, and water turbidity levels would probably have been very high which combined with overcast conditions when barely daylight (if the fish did enter the tunnel near low water) would have meant conditions of minimum visibility for the fish. This, combined with the turbulent state of the sea, may have been partly responsible for fish entering the inlet. The combination of circumstances at the time would obviously be only the final link in a chain which initially would require a high general level of shoal abundance in the vicinity. The pattern of wind flow in the days preceding may then have produced an abnormally high density of fish close to the shore within the tidal flow-line of the inlet.

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5	10/5/2012	JM	BJR	Prel B	Clarifies relationship with previous report TR065 – no other changes	
4	22/09/11		PJ	BPE/VSOSR	Minor comments to be addressed in a future version	CJLT
4	15/09/11		PJJ	PREL A	Report de-restricted at EDF request	
3 (Ed 2)	30/03/11	JDM	AP	PREL A	Ed 2 with effect of AFDs added and statistical analysis of community structure	
2	21/03/11	JDM	BJR	BPE	Minor amendments addressed	MH
1	15/02/11	JDM	BJR	PREL B	Clarifications after EDF comments	
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A synthesis of impingement and entrainment predictions for NNB at Hinkley Point

Not Protectively Marked

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Executive summary

This report presents a synthesis of the impact of predicted impingement and entrainment rates for the planned new nuclear build at Hinkley Point (HP C) as at April 2011. Results are presented for:

a) The existing HP B reactor with a cooling water intake of $33.7 m^3 s^{-1}$
b) A traditional design of HP C with no impingement mitigation measures and the planned cooling water intake of $125 m^3 s^{-1}$
c) The planned HP C design fitted with proposed mitigation features of acoustic fish deterrence (AFD), low-velocity intakes and a fish recovery and return (FRR) system

The HP B results enable the HP C predictions to be put into context of both the local commercial fishing activities and the local fish populations. The results also allow the combined effects of operating HP B+C together to be assessed should the life of HP B be extended.

This report differs from an earlier report, "**Predictions of impingement and entrainment by a new nuclear power station at Hinkley Point, Edition 2**" (BEEMS Technical Report TR065) in the following manner:

a) The calculations incorporate corrections to the HP B Comprehensive Impingement Monitoring Programme (CIMP) dataset that were discovered after TR065 Edition 2 was finalised.
b) Calculations of HP B impingement are based upon actual rather than nominal cooling water flow rates (33.7 *vs.* 30 cumecs).
c) Includes the predicted effect of the proposed HP C impingement mitigation measures (acoustic fish deterrents (AFD) and fish recovery and return (FRR) systems).
d) It provides more detailed assessments for certain conservation species (particularly eel, shad and lampreys) in relation to population abundance.
e) It takes account of a simplified approach to estimating entrainment

This report is a synthesis document and it therefore does not include all the detail (e.g. methodological and fisheries information) originally presented in TR065. Furthermore, there are components of TR065 (e.g. the output from PISCES 2009) that are now redundant as there is sufficient survey data available from the CIMP.

The previous estimates of impingement and entrainment presented in TR065 are therefore now superseded by those presented in this report. However the reader is referred to TR065 for more extensive methodological detail where appropriate.

Impingement

Predictions are based upon scaling up the results of the Comprehensive Impingement Monitoring Programme (CIMP) carried out at HP B over 12 months from February 2009 to February 2010 (BEEMS Technical Report TR129). Where suitable and appropriate biological data are available, these predictions have been put into the context of local commercial landings and local fish populations (spawning-stock biomass, SSB).

The intakes at HP B are 550m offshore, whereas those planned for HP C are 3.3km offshore. The question then arises of how comparable the fish populations are at these inshore and offshore locations. The results of an extensive series of fishing surveys at Hinkley Point using different gears are presented in BEEMS Technical Report TR083. This report concluded, that from the available data, it was not possible statistically to differentiate between the fish populations (in species or numbers) in the vicinities of the two intake structures. Without mitigation, impingement at HPC will therefore increase about fourfold over that of HPB simply as a result of the increased abstraction of cooling water (impingement rates of the different species differ, of course).

The Environment Agency Best practice guide for intake screening (Turnpenny and O'Keefe 2005) recommends that the following measures should be implemented to reduce impingement:

1. That intakes should be designed to reduce face velocities to a target flow rate of $0.3ms^{-1}$.
2. That a suitable fish deterrent should be fitted at the intakes that will work synergistically with the low velocity intake.
3. That a fish recovery and return system should be fitted to improve the survival rates of those fish that are impinged.

BEEMS Technical Reports TR117 and TR065 describe the design of the proposed intake structures for HP C and the potential impact upon impingement rates.

It is important to emphasise that these predictions are predicated upon the fish being able to detect and respond to the intake structure. At the high turbidities usually present at Hinkley Point, this would depend in the absence of visual cues upon the presence of suitable stimuli, e.g. turbulence generated at the periphery of the intake structure by screen slats or by artificial stimuli generated by acoustic fish deterrents.

BEEMS Technical Report TR181 presents an investigation into the likelihood that louvre screens at the HP C inlets would provide the required stimuli to enable fish to detect the structures. The report concluded that there is insufficient evidence to be able to predict any likely fish guidance efficiency of such louvre screens, or whether they would meet EA best available technology (BAT) requirements.

In order to meet BAT requirements for impingement mitigation EDF have commenced a design study to investigate the issues surrounding fitting AFDs near the HP C intakes. The predicted impact of using AFDs and low velocity intakes is presented in this report. Given that no site surveys of impingement are available for this design of cooling water intake, it is impossible to state with certainty the extent that the estimated reduction in impingement will be realised. It is therefore important to carry out impingement monitoring at Hinkley Point C once after it is in operation to determine (1) the effectiveness of the new design in reducing impingement mortality and (2) any need for additional fish deterrent systems.

It should also be noted that low-velocity water intakes combined with AFDs are unlikely to affect impingement rates for species that are weak swimmers or that have relatively poor hearing such as eel, lamprey and crustaceans. For those species, in the absence of a fish recovery and return (FRR) system, the impingement predictions based upon a conventional intake design are likely to be the most realistic.

Implementation of a FRR system will further reduce the mortality impact of impingement for many species and the combined effect of a low-velocity intakes, AFDs and FRR could result in impingement mortalities that are, for many species, not dissimilar to the current situation at HPB if the full potential of these three mitigations are realised.

Impingement Predictions

The proposed HP C has been specified with low velocity intake structures, AFDs and a Fish Recovery and Return system. If these proposed impingement mitigation measures function as designed, the impingement losses at HP C are calculated to be similar to those of the existing HP B station. The resulting HP C impingement losses will have a negligible effect on the spawning stock of the protected migratory species that use the Severn Estuary and have been captured on the intake screens of HP B (European eel (*Anguilla anguilla*), sea lamprey (*Lampetra fluviatalis*) and twaite shad (*Alosa fallax*). Both the shads are pelagic species, and although limited biological and stock information is available from the literature, we know that they, as all pelagic species, are subject to massive fluctuations in recruitment. These annual fluctuations are related, generally, to environmental stimuli, either directly or indirectly, and can, for relatively small stocks, be as much as an order of magnitude. Given that level of annual variability, our considered opinion is that an annual 0.5% loss in twaite shad stock as a result of the operation of HPC is not significant and would not have an adverse affect upon the integrity of the SAC based on our scientific understanding of the dynamic nature of such pelagic stocks. The catches of Allis shad and Salmon on the HP B input screens are too small to allow a reliable impingement loss to be calculated. No Sea Trout (*Salmo trutta*) have been caught during any impingement sampling at Hinkley Point.

The impact on the commercially important fish species that represent the majority of the existing impingement losses (Sprat, Whiting, Sole, Plaice, Herring and Blue Whiting) is considered to be negligible. For Whiting, Sole, Plaice and Blue Whiting the impingement losses will have a negligible effect on the spawning stock. Sprat is the dominant (>97%) clupeiform fish impinged at HP B and the population trend for this group since 1981(Figure 1) has remained stable. As HP C (with mitigation) will only impinge about half the number of sprat compared to the current HP B, we conclude that HP C (with mitigation) is unlikely have any significant impact on local sprat population. For Herring the impingement losses are about 0.2% of the local fishery and will therefore have negligible impact on the local population. The impact on cod would represent 3.24% of the local SSB. This level of loss is equivalent to 1% of the Total Allowable Catch of cod recommended by ICES for 2011 for Divisions VIIe-k (3420 t) and is unlikely to have any measurable effect on the local cod population when considered against the background natural variability in SSB. For whiting (*Merlangius merlangus*) and cod (*Gadus morhua*) the predicted impingement is comparable with local fish catches but it should be noted that the commercial fishery for these species in the area is very small. The predicted impingement losses on crustaceans (as represented by the impact on the brown shrimp *Crangon crangon* the main crustacean impinged) are also expected to be similar to those of HP B. BEEMS Technical Report TR071 Edition 3 provides further information on the fishery in the vicinity of the proposed NNB.

Five fish species of conservation concern – Atlantic salmon, European eel (*Anguilla anguilla*), river and sea lampreys (*Lampetra fluviatilis, Petromyzon marinus*) and twaite shad (*Alosa fallax*) – migrate through the Severn Estuary and also have been caught in very small numbers on the cooling water input screens at Hinkley Point B (although there have been no salmon in the past five years). Predictions of impingement are provided for these species, together with an assessment of the ecological significance of such losses.
The predicted impingement losses on crustaceans (as represented by the impact on the brown shrimp *Crangon crangon* the main crustacean impinged) are also expected to be similar to those of HP B.

Entrainment

The community of fish eggs and larvae at Hinkley Point is small in both species and numbers and the predicted entrainment losses are insignificant for those species for which we have been able to make an assessment. The impact of entrainment on the shads and lampreys that spawn and live as larvae in the freshwater tributaries of the Severn is expected to be negligible.

Within the February–June spawning period, the numbers of eggs and larvae of sea bass entrained by a NNB at Hinkley Point are predicted to be <0.45% of the mean abundance within the Trevose spawning ground. This reduces to ~0.1% when survival of eggs and larvae as estimated from EMU studies are taken into account. For sole and sprat the numbers of entrained eggs and larvae over the same period are predicted to be <0.005% of the mean abundance within the Trevose spawning ground. However, this entrainment prediction should be used with caution because several assumptions have been made in the estimation and interannual variability (which is likely to be great) could not be taken into account.

Impacts of impingement and entrainment on the local fish community

An analysis of the abundance trends by species group from 1981 to 2008 from the long-term impingement monitoring programme dataset for Hinkley Point collected and collated by Pisces Conservation Ltd shows that HP B has not had any obvious positive or negative effect on the fish community structure at Hinkley Point. Where the data do indicate some population changes (e.g. for European Eel) these would appear to have been the consequence of a change in species abundance across the northeast Atlantic broadly rather than losses to impingement or entrainment at Hinkley Point B itself.

Table of contents

1. Introduction 15

2. Predicting entrainment 16

2.1 Mortality of entrained organisms 18

3. Predicting impingement 19

3.1 Low-velocity intake and Acoustic Fish Deterrent mitigation 19

3.2 Fish Return and Recovery mitigation 20

4. Predictions and impacts of entrainment by species 22

5. Predictions and impacts of impingement by species 25

Crustaceans 25

Sprat 26

Whiting (BAP) 27

Sole (BAP) 28

Cod (BAP) 29

Herring (BAP) 30

Plaice (BAP) 30

Blue whiting (BAP) 31

Eel (Eel management plan) 32

Twaite shad (SAC designated) 34

Allis shad (SAC designated) 34

Lamprey (SAC designated) 35

Salmon (SAC designated) 36

6. Impacts of impingement and entrainment on the local fish community 37

7. Overall Conclusions 41

7.1 Predicted effects of HP C based upon HP B monitoring data 41

7.2 Predicted combined effects of HP C+ HP B 42

8. References 43

Appendix A. Estimated total annual Impingement for HP B and predicted total annual Impingement for HP C by species. 46

Appendix B1. Estimated total annual impingement for HP B for selected species assuming an abstraction rate of 30 cumecs via current intake structures compared with local fishery and estimated local population size. 50

Appendix B2. Predicted total annual impingement for HP C for selected species assuming an abstraction rate of 125 cumecs via current intake structures compared with local fishery and estimated local population size. 50

Appendix B3. Predicted total annual impingement for HP C for selected species assuming an abstraction rate of 125 cumecs via current and low velocity intake structures compared with local fishery and estimated local population size. 50

Appendix B4. Predicted total annual impingement for HP C for selected species assuming an abstraction rate of 125 cumecs via current and low velocity intake structures and with a fish recovery and return system compared with local fishery and estimated local population size. 50

Appendix C Trend Analysis of the long term HP B fish Impingement data 55

List of Tables and Figures

Tables

Table 1. Historical EMU mortality results. Summary of zooplankton and ichthyoplankton survival results obtained using EMU. These results are based on ΔT 10°C and a chlorine concentration of 0.2 ppm (BEEMS Technical Report TR081).

Table 2. Predicted total annual impingement (numbers of fish, EAV) at Hinkley point C and Hinkley Point B for selected species assuming an abstraction rate of 125 cumecs via current intake structures and via low-velocity intake structures with AFD and with a fish recovery and return system. (Data from Appendices B1, B2, B3 & B4)

Table 3. Predicted total annual impingement (numbers of fish) at Hinkley point C for selected species assuming an abstraction rate of 125 cumecs via current intake structures and via low-velocity intake structures with AFD and with a fish recovery and return system compared with local fishery and estimated local population size. ("NA" indicates no assessment made, data from Appendix B4).

Table 4. Presence (+) of fish eggs and larvae detected in Ichthyoplankton surveys off Hinkley Point in 2008 and 2009.

Table 5. Predicted entrainment of fish eggs and larvae between February and June at Hinkley Point C (based on the BEEMS 2010 plankton survey) in relation to the abundance in the Trevose spawning area.

Table 6. The 13 most abundant species detected in the routine long-term impingement monitoring programme during the periods 1985-1989 and in 2004-2008.

Table 7. Differences in the presence/absence of fish species detected in the routine long-term impingement monitoring programme between the periods 1985-1989 and 2004-2008.

Table 8. The composition of the species groups used in the analysis of abundance trends from 1981 to 2008.

Figures

Figure 1. Abundance trends by species group and year at Hinkley Point B from 1981 to 2008 derived from the long-term impingement monitoring programme dataset (Pisces Conservation Ltd). Annual abundance (vertical axis) is expressed as: ln(1+mean monthly catch).

1. Introduction

Like other coastal power stations with 'once through' cooling systems, Hinkley Point B power station abstracts large volumes (~30+ $m^3 s^{-1}$) of seawater to condense the turbine steam. The new nuclear build (NNB) at Hinkley Point C will also consist of a 'once through' cooling system design, although the volumes of cooling water abstracted are likely to be larger (~125 $m^3 s^{-1}$). Although the cooling water intakes will be protected by coarse louver screens to prevent the intake of larger fish and debris, a significant number of small organisms (small fish and crustaceans, and plankton) inevitably enter the cooling water intake. The larger organisms must be removed before the water enters the power station cooling system to prevent them blocking the condenser tubes. These organisms (fish and crustaceans >25 mm length) are removed through impingement on fine-mesh (currently 10 mm, but 5-mm is planned for the NNB) drum screens. The smaller organisms (mostly the eggs and larvae of fish and crustaceans) that pass through the drum screens are entrained and pass through the power station cooling system without causing significant blockages.

Despite some power stations (not currently Hinkley Point B, but planned for the NNB) having fish-return systems that collect fish from the drum screens and return them to sea, a varying proportion of the animals impinged by the cooling water circuit ultimately die, either as a direct consequence of the impingement process or subsequently as a result of a likely increased susceptibility to predation and disease after returning to the sea.

In an initial report (BEEMS Technical Report TR065, ("*Initial predictions of impingement and entrainment by a new nuclear power station at Hinkley Point*"), we provided preliminary estimates of likely impingement and entrainment of fish and crustaceans by species in relation to the fisheries and fish populations local to Hinkley Point. These predictions were based primarily on data derived from 1) the routine, impingement monitoring programme that has been underway at Hinkley since the early 1980s, and 2) a computer-based predictive tool (PISCES 2009), updated as part of the BEEMS programme. These two sources of data were the only ones available at the time, but neither represented recent, unbiased, comprehensive data for actual impingement at Hinkley, so BEEMS initiated a Comprehensive Impingement Monitoring Programme (CIMP) there. Because the complete results from the CIMP were not available for the first edition of TR065, we undertook to revise TR065 once the CIMP was complete: a second edition has now been completed ("*Predictions of impingement and entrainment by a new nuclear power station at Hinkley Point, Edition 2*").

By their nature, both editions of TR065 are long, drawing on a range of different and extensive datasets as well as other information, some of which only became available between drafting the 1st and the 2nd editions. In addition, new information has recently become available through the BEEMS project about further potential mitigation measures (e.g. fish-return systems) that can be used to refine predictions. While TR065 provides a comprehensive assessment of impingement and entrainment at a NNB at Hinkley, we are aware that there is value in preparing a short, simple updated synthesis.

Here we summarise the predicted impingement impact of HPC on individual CIMP species, the predicted impact on relevant local populations of the major commercial species and designated conservation species, and the predicted impact on local community structure. To achieve this we have used unified output predictions from TR065 (2nd edn) for impingement and entrainment, but have extend the estimate Equivalent Adult Value for selected species (see below). We have also assessed the impacts of low-velocity intake mitigation (using BEEMS Technical Report TR117 as source data) and of Fish Return and Recovery mitigation (using BEEMS Technical Report TR106 as source data). Where the necessary biological information is lacking, we provide our best approximation together with an explanation of the caveats that should append our assessment.

We also provide best estimates for entrainment, given available data – filling gaps with analogues as necessary, using published entrainment mimic unit (EMU; see 2.1 below) survival data where available, and identifying species of conservation and commercial interest that are less vulnerable to entrainment.

This report differs from BEEMS Technical Report TR065 in the following manner:

a) The calculations incorporate corrections to the HP B Comprehensive Impingement Monitoring Programme (CIMP) dataset that were discovered after TR065 Edition 2 was finalised.
b) Calculations of HP B impingement are based upon actual rather than nominal cooling water flow rates (33.7 *vs* 30 cumecs).
c) Includes the predicted effect of the proposed HP C impingement mitigation measures (acoustic fish deterrents (AFD) and fish recovery and return (FRR) systems).
d) It provides more detailed assessments for certain conservation species (particularly eel, shad and lampreys) in relation to population abundance.
e) It takes account of a simplified approach to estimating entrainment

This report is a synthesis document and it therefore does not include all the detail (e.g. methodological and fisheries information) originally presented in TR065. Furthermore, there are components of TR065 (e.g. the output from PISCES 2009) that are now redundant as there is sufficient survey data available from the CIMP.

The previous estimates of impingement and entrainment presented in TR065 are therefore now superseded by those presented in this report. However the reader is referred to TR065 for more extensive methodological detail where appropriate.

2. Predicting entrainment

Entrained material typically includes holoplanktonic organisms (permanent members of the plankton, such as copepods, diatoms and bacteria) and meroplanktonic organisms (temporary members of the plankton, such as shrimps and the planktonic eggs and larvae of invertebrates and fish) (Bamber *et al*., 2004).

Of those species likely to be entrained in the cooling water intake at Hinkley Point, the most important/abundant ichthyoplankton species will be gobies and flatfish, and the most important/abundant zooplankton species will be mysids.

It is possible to estimate the numbers of organisms lost to entrainment during operation at Hinkley Point with a cooling water extraction of 125 $m^3 s^{-1}$ using existing BEEMS, Cefas and published data if we make two assumptions: first, that larvae and eggs are drawn in volumetric proportion to their presence in surrounding waters (ICES statistical rectangles 29E4, 30E4, 31E4, 30E5, 31E5 and 31E6); and second, that there is 100% mortality of entrained organisms. Neither of these assumptions is likely to be true. The BEEMS entrainment feasibility study conducted during 2008, albeit only for a few hours of sampling, observed higher densities of ichthyoplankton in the intake samples than at the coastal site at Hinkley Point B. This contrasts with the study by Dempsey (1988) at Fawley Power Station, where the densities of larvae in Southampton Water were greater than those entrained from the entire water column, indicating that fish larvae were able to avoid entrainment and that actual entrained numbers were significantly lower than would be expected from offshore plankton surveys.

With respect to the impact of mortality of eggs and larvae on fish populations through entrainment of ichthyoplankton at power stations, we consider the estimation of Equivalent Adult Value (EAV) using the method described by Turnpenny (1989; see later) to be problematic. Estimates for the biological parameters are not as readily available in the way they are for older fish. The rate of natural mortality of eggs is difficult to estimate, because it varies by development stage, temperature (duration of egg stages) and the abundance of predators (at least), and the survival of larvae also depends on the availability of food. Therefore, linking numbers of eggs and larvae to fecundity is prone to high levels of uncertainty and the precision of EAV estimates will be correspondingly low. Given these uncertainties, and in the absence of data for many of the parameters required, we have estimated the impact of entrainment of fish eggs and larvae at Hinkley Point from the filtering power of the intake and ichthyoplankton densities local to Hinkley Point derived from intensive monthly plankton surveys conducted between February and June 2010 (See BEEMS Technical Report TR083a)*.

Ichthyoplankton varies spatially throughout the Bristol Channel, being highest for eggs in the spawning areas (particularly around Trevose Head, for most commercial species), and may also be high nearshore where larvae and post-larvae begin to recruit to nursery areas (e.g. for sea bass, see Jennings and Pawson, 1992). In this respect, the water filtered at Hinkley Point will not be representative of other areas of the Bristol Channel, although the inner reaches of the Severn Estuary are well mixed.

For certain species, the predicted entrainment of eggs and larvae at Hinkley Point can be estimated from the filtration value weighted by the estimated abundance of eggs and larvae in the intake water compared with that in the main production areas of eggs and larvae.

Given the above limitations we provide a prediction of entrainment at Hinkley Point based on:

- identifying the body of water in the vicinity of the proposed intake and estimating the volumetric turnover by a NNB pumping 125 $m^3 s^{-1}$
- identifying plankton abundance within this water body
- assuming no stratification of plankton in the water column and that the intake is not selective, larvae and eggs are drawn in volumetric proportion to their presence in surrounding waters
- assuming 100% mortality during passage through the cooling water circuit, except for species where EMU studies indicate otherwise (Dover sole and bass, see Table 1).

2.1 Mortality of entrained organisms

Bamber *et al*. (1994) described an entrainment mimic unit (EMU) designed to mimic realistically the conditions of entrainment passage through the cooling-water system of a coastal power station under laboratory conditions as a means of assessing likely mortalities of entrained organisms. The apparatus allows the assessment of the effects of the four key stressors of entrainment: temperature, pressure, biocide and mechanical effects, alone and in combination. Their original experiments on larvae of the Pacific oyster (*Crassostrea gigas*) gave a baseline comparison of the technique to a standard bioassay technique (the D-stage larval test) and demonstrated the suitability of the apparatus and experimental protocols to assess the impacts of power-station entrainment.

Of the species tested in previous EMU studies, three [sole (*Solea solea)*, turbot (*Psetta maxima*) and the brown shrimp (*Crangon crangon*)] are likely to be entrained at Hinkley Point. For example, Bamber and Seaby (2004) concluded that, in combination, the stresses of entrainment under standard power-station operating levels would result in approximately 20% mortality of brown shrimp larvae (from the combination of total residual oxidant (TRO), and rise in temperature (ΔT). However, for the purpose of the current report, we are hesitant to apply this or other EMU survival rates detailed in Table 1 to Hinkley Point, because the pressure profiles used in the EMU were all based on those for Sizewell B. Nonetheless, it would appear that survival rates for fish eggs may be quite high (80+%), and that survival rates for fish larvae are lower and more variable (Table 1).

Table 1 Historical EMU mortality results. Summary of zooplankton and ichthyoplankton survival results obtained using EMU. These results are based on ΔT 10oC and a chlorine concentration of 0.2 ppm (BEEMS Technical Report TR081).

Species	% Survival at life stage	% Survival at life stage	% Survival at life stage
	Eggs	Larvae	Adults
Sole	90	-	–
Turbot	93	27	–
Sea bass	80	70	–
Brown shrimp	–	73	–
Lobster	–	85	–
Copepod	–	–	80
Mussel	–	47	–
Pacific oyster	–	5	–
Eel	–	52	–

3. Predicting impingement

Although three, quasi-independent, sources of impingement were used to predict the levels of impingement at a NNB at Hinkley Point C for TR065, predictions in this report are primarily based on a Comprehensive Impingement Monitoring Programme (CIMP) carried out over 12 months from February 2009 to February 2010 (see above). For a few species, where suitable and appropriate biological data are available, these predictions have been put into the context of local commercial landings and local fish populations (spawning-stock biomass, SSB).

Data from CIMP are presented for up to 64 species of fish and up to 14 species of crustacean (Appendix A). For many of these species the predicted impingement is based upon very small numbers of individuals caught on the screens of existing power stations during limited (40 x 24 h) sampling intervals at an abstraction rate of 30 $m^3 s^{-1}$ (cumecs). The predicted impingement has been calculated by scaling the numbers up to a full year at the proposed cooling water abstraction rate of 125 $m^3 s^{-1}$. For example, only two Allis shad (*Alosa alosa*) were caught, but after scaling up, this leads to a predicted impingement of 68 individuals per year. Such impingement predictions for species caught infrequently are subject to considerable uncertainty, so must be treated with caution.

In this report, the CIMP data (above) have been rescaled to proved estimates of annual impingement assuming full cooling water pumping capacity for:

1. The current Hinkley Point B station pumping 33.7 $m^3 s^{-1}$ (cumecs) (Appendix A).
2. Hinkley Point C station pumping 125 $m^3 s^{-1}$ (cumecs) assuming current intake location and configuration (Appendix A).

For some (13) species of commercial and/or conservation importance, sufficient data are available to make an assessment of the impact of predicted impingement on the local fish populations in the Severn Estuary area. For those species, we have further refined the assessments to identify the potential reductions in impact that could possibly be achieved by implementing 1) low-velocity intake and acoustic fish deterrent mitigation (using TR117 as a source of data and methods) and 2) fish return and recovery mitigation (Appendices B1 to B4 and Tables 2 & 3). Full details of the methods and source data are given in TR065, 2nd Edition.

3.1 Low-velocity intake and Acoustic Fish Deterrent mitigation

The key to reducing fish impingement is to design the cooling-water intake structures for a maximum intake velocity that will allow most fish to escape, and the Hinkley Point C project has a design target of ≤0.3 $m s^{-1}$. Earlier studies (O'Keeffe and Turnpenny, 2005) showed that this would allow larger individuals of most coastal fish species to avoid entrapment, provided that they also have appropriate sensory cues to encourage avoidance behaviour.

Because of the usual high water turbidity at Hinkley Point and the consequent absence of visual clues, any mitigating effect of the low-velocity intake is only likely to be realised if it is combined with some form of artificial stimulus (e.g. an acoustic fish deterrent) to induce fish to swim away from the intake structure. Equally however, an acoustic fish deterrent is unlikely to be fully effective on its own if the intake velocity exceeds the swimming capabilities of the fish. For these reasons low-velocity intake and AFD need to be considered as a combined mitigation measure.

The type of AFD system most suitable for estuarine power station cooling water intakes uses a sound projector array (SPA) that consists of an array of underwater loudspeakers or "sound projectors" that produce a diffuse field that will repel the entry of hearing-sensitive fish. While many types of fish are repelled by AFD systems, deflection efficiency tends to be lowest (typically <30%) for benthic and epibenthic species (e.g. flatfishes) that lack a fully developed swimbladder, higher for species with a swimbladder, and highest (>80%) for species that have well developed hearing, e.g. clupeids such as herring, sprat and shad.

Although there are few published studies of the effectiveness of acoustic fish deterrents at power station cooling water intakes, one such study (Maes et al. 2004) conducted at the Doel nuclear power station on the Scheldt Estuary in Belgium provides relevant information for some of the more important species found at Hinkley Point. However, this source does not consider the effects of AFDs at different intake velocities. Given that it is intended that Hinkley Point C will have a low-velocity intake, we have assumed for the purposes of this report that it is the AFD that will have the predominant impact in reducing entrapment and we have therefore used the data from Maes et al (2004) to estimate (below and in Appendix B3) the reduction in entrapment that is likely to result from combined AFD and low-intake velocity at Hinkley Point C for individual species.

Although we have assumed combined AFD and low-intake velocity, the calculations do not include the reductions in impingement estimated to occur as a consequence of the low-velocity intake alone as presented in TR117 (Assessment of Effects of CW Intake Velocity on Fish Entrapment Risk at Hinkley Point). Maes et al. (2004) suggested that the AFD at Doel was more effective in deflecting larger fish because, having a higher swimming speed, they were better able to escape the intake than smaller fish. Therefore, because an AFD affects the size/frequency distribution of the fish impinged, increasing the relative proportion of smaller fish, the estimates of impingement reduction presented in TR117 are not appropriate because they are based on impingement size/frequency data recorded at Hinkley in the absence of an AFD.

3.2 Fish Return and Recovery mitigation

The use of direct water cooling by power stations is accepted best available technology (BAT) under the European Integrated Pollution Prevention and Control (IPPC) Directive, but is questioned on the grounds of environmental harm by some commentators and has largely been phased out in the USA. However, since publication of the supporting BAT reference advice in 2000, there have been marked advances in intake mitigation techniques that can substantially reduce abstraction-related impacts. These include the use of fish recovery and return (FRR) techniques and current Environment Agency best practice guidance on intake mitigation advises the use of FRR techniques for estuarine and coastal abstractions.

FRR techniques have been used at a number of sites in the UK and overseas, and the recent upsurge in environmental regulation emanating from the European Commission has led to increased interest in FRR systems and has spurred their further development. However, the form and level of sophistication of the installations range widely, with corresponding variation in effectiveness and few conform to current EA standards of Best Practice.

Currently, most FRR survival studies have been relatively short-term (typically 24-96 h), and few have considered the mid- to long-term effects on the susceptibility to disease or predation. Well-designed FRR systems can achieve 80-100% survival rates for more robust epibenthic species like plaice and flounder, and moderate rates (~50-60%) for demersal species such as the robust gadoids (e.g. cod). However, survival rates for delicate pelagic species like herring, sprat and shad are usually low (<10%). It is also important to note that gadoids and other physoclistous (closed swimbladder) species are prone to pressure-related swimbladder damage in systems where cooling water intakes are 10 m or more below the water surface. The proposed cooling water intake for Hinkley C is at -12.4 m ODN (Ordinance Datum Newlyn). The design is such that the intakes are approximately 1.5 m clear of the seabed and the inlet aperture ~2m high. Therefore, the mean depth of the intake is about -10 m ODN. However, as the tidal range at Hinkley is 12 m on Spring tides (± 6 m), the water depth from which the fish are extracted will vary from 4 m to 16 m deep, and this is likely to reduce the effectiveness of an FRR for physoclistous species such as cod.

It must also be noted that the use of low-velocity cooling water intakes (above) will significantly affect the size and age composition of the fish that are impinged on the intake screens, with smaller and younger fish dominating, and any differences in survivorship between small and large individuals of the same species may impact the overall effectiveness of FRR systems.

The proposed FRR system for HP C is being designed to achieve high rates of survival for eels and lamprey in particular, but it is expected that rates for other epibenthic and demersal species will also be higher than achieved in older designs. However, for the purpose of this study we have assumed the following conservative recovery rates taken from the EA best practice guidance (O'Keeffe and Turnpenny, 2005):

Group	**Survival rate**
Pelagic (e.g. herring, sprat, shad)	0%
Demersal (e.g. cod, whiting, gurnards)	50%
Epibenthic (e.g. flatfish, eels, gobies, rocklings and crustaceans)	80%

We have included our assessment of the potential impacts of fish recovery and return mitigation for individual species below.

Table 2 Predicted total annual impingement (numbers of fish, EAV) at Hinkley point C and Hinkley Point B for selected species assuming an abstraction rate of 125 cumecs via current intake structures and via low-velocity intake structures with AFD and with a fish recovery and return system. (Data from Appendices B1, B2, B3 & B4)

Species: common name	HPC, current intake design.	HPB	HPC with low-velocity intake & AFD (increase from current HPB)		HPC with low-velocity intake, AFD & FRR (increase from current HPB)	
Sprat (largest numbers)	3,380,850	936,386	405,702	-(57%)	405,702	-(57%)
Whiting (BAP)	288,078	79,253	129,635	(64%)	64,818	-(18%)
Sole (BAP)	32,429	8,559	27,241	(218%)	5,448	-(36%)
Cod (BAP)	32,063	8,733	14,428	(65%)	7,214	-(17%)
Herring (BAP)	44,792	12,570	2,240	-(82%)	2,240	-(82%)
Plaice (BAP)	493	129	414	(221%)	83	-(36%)
Blue whiting (BAP)	160	46	72	(55%)	36	-(22%)
Eel (Eel management plan)	1,304	351	1,304	(272%)	261	-(26%)
Twaite shad (SAC designated)	2,276	646	273	-(58%)	273	-(58%)
Allis shad (SAC designated)	68	22	8	-(63%)	8	-(63%)
Sea lamprey (SAC designated)	207	42	207	(398%)	41	(0%)
River lamprey (SAC designated)	82	18	82	(355%)	16	-(9%)
Salmon (SAC designated)	0	0	0	(0%)	0	(0%)
Sea Trout (SAC designated)	0	0	0	(0%)	0	(0%)
Crangon crangon (the main crustacean impinged)	19,135,756	4,911,592	19,135,756	(0%)	3,827,151	-(22%)

Table 3 Predicted total annual impingement (numbers of fish) at Hinkley point C for selected species assuming an abstraction rate of 125 cumecs via current intake structures and via low-velocity intake structures with AFD and with a fish recovery and return system compared with local fishery and estimated local population size. ("NA" indicates no assessment made, data from Appendix B4).

Species: common name	EAV (number, current intake)	EAV (number, low-velocity intake, AFD & FRR)	local fishery (t)	local SSB (t or number)	AFR &FRR: %of local fishery	AFR & FRR: % local SSB
Sprat (largest numbers)	3,380,850	405,702	0.19	NA	1665.5%	-
Whiting (BAP)	288,078	64,818	33.5	1613	34.4%	0.72%
Sole (BAP)	32,429	5,448	263	3240	0.5%	0.04%
Cod (BAP)	32,063	7,214	65.2	975	48.5%	3.24%
Herring (BAP)	44,792	2,240	119.4	NA	0.2%	-
Plaice (BAP)	493	83	84	952	0.0%	0.00%
Blue whiting (BAP)	160	36	37,900	5,360,000	0.0%	0.00%
Eel (Eel management plan)	1,304	261	-	133.4	-	0.06%
Twaite shad (SAC designated)	2,276	273	-	184,000	-	0.15%
Allis shad (SAC designated)	68	8	-		-	
Sea lamprey (SAC designated)	207	41	-	15,269	-	0.27%
River lamprey (SAC designated)	82	16	-	116,109	-	0.01%
Salmon (SAC designated)	0	0	0	0	0	0
Sea trout (SAC designated)	0	0	0	0	0	0
Crangon crangon (the main crustacean impinged)	19,135,756	19,135,756	-	-	-	-

4. Predictions and impacts of entrainment by species

Ichthyoplankton surveys off the Hinkley Point area were undertaken quarterly in 2008 and again in May 2009 (BEEMS Technical Report TR083). Eggs and larvae of just 14 species were detected in very low numbers (Table 4 below shows which species were detected during 2008/9). However, those surveys were designed to increase understanding of the subtidal ecology of the area and not just the ichthyoplankton community, so the timing of the surveys in 2008 were not optimal for the main fish spawning season.

Table 4 Presence (+) of fish eggs and larvae detected in Ichthyoplankton surveys off Hinkley Point in 2008 and 2009.

Species	Eggs	Larvae
Anchovy	+	
Dover sole	+	+
Rockling spp	+	
Solonette	+	+
Sea bass	+	+
Gurnard spp	+	
Dragonet		+
Herring		+
Sprat		+
Sandeel		+
Goby spp		+
Mackerel	+	
Pilchard	+	
Scaldfish	+	

In order to obtain a better estimate of ichthyoplankton communities at the site intensive monthly surveys were undertaken between February and June 2010 (BEEMS Technical Report TR083a). Despite this greatly increased sampling effort, the eggs and larvae of only 18 species were detected, although much better temporal and spatial density estimates were obtained. The 2010 surveys confirmed the findings of the 2008 and 2009 surveys that the Hinkley Point area has a very limited ichthyoplankton community and therefore the risk of entrainment loss is both low and is limited to a narrow range of species. A few comments are made in BEEMS TR083a about the likely source of the eggs and larvae identified during the ichthyoplankton surveys. Succinctly, the conclusion is that there is very little, if any, local spawning of the species listed. Moreover, fishing activity quickly identifies notable spawning activity of commercially important fish, and BEEMS TR071 V 03 clearly shows that the area is not a great interest to fishers, even if minor spawning events occur nearby.

Although eggs and larvae of 18 species of fish were detected in the BEEMS intensive plankton survey off Hinkley Point in 2010 (TR083a), comparison with abundances at the Trevose spawning area have only been made for European sea bass, Dover sole, and sprat because these are the only ones of commercial interest identified during the BEEMS plankton surveys that can be compared with those species present.

Table 5 Predicted entrainment of fish eggs and larvae between February and June at Hinkley Point C (based on the BEEMS 2010 plankton survey) in relation to the abundance in the Trevose spawning area.

Predicted entrainment of fish eggs and larvae at Hinkley Point C (based on the BEEMS 2010 plankton survey) in relation to the abundance in the Trevose spawning area.					
Species/species group	**Eggs**	**Larvae**	**A: Total§**	**B: Trevose**	**A/B (%)**
Sandeels		9,075,949	9,075,949		
Solenette	368,278	2,496,257	2,864,536		
Five-bearded rockling		333,687	333,687		
Herrings		414,615	414,615		
European sea bass	47,282,931	41,981,786	22,051,122	20,906,261,000	0.11%
Rocklings	18,546,479	799,420	19,345,899		
Gobies		10,351,234	10,351,234		
Butter fish		389,819	389,819		
European Flounder		2,711,333	2,711,333		
European plaice		3,322,735	3,322,735		
Pilchard	2,891,002	386,310	3,277,311		
Dover sole	9,461,839	1,929,208	1,659,991	274,633,000,000	0.001%
Soles*	450,281	369,308	819,589		
Sprat		7,114,303	7,114,303	478,943,000,000	0.001%
Sea scorpion		474,262	474,262		
Unidentifiable fish	5,004,020	21,332,227	26,336,246		
European anchovy	12,141,963		12,141,963		
Dragonets	383,685		383,685		
§ For Dover sole and bass, the results have been adjusted as to account for estimated survival based on EMU experiments (Table 1)					

**"soles" indicates eggs and larvae that could, by dint of damage, could not be confirmed as Dover sole, but were identified as belonging to the family Soleidae.*

The estimated entrainment of eggs and larvae over the period February–June given in Table 5 has been made assuming:

- no exchange between the pool and adjacent sea areas;
- uniform distribution and abundance of ichthyoplankton throughout the water column;
- the mean ichthyoplankton abundances from the 2010 surveys close to Hinkley Point power station occur within the identified 'pool'.

These entrainment estimates can be compared and put into context with the abundance of ichthyoplankton at the Trevose Head ground by examining the mean abundance of the same species in the Trevose spawning area (Horwood & Greer-Walker, 1990), ICES rectangles 29–31E4, 30–31E5 and 31E6, assuming that:

- the mean abundances of eggs and larvae from the 1990 surveys were within the ICES rectangles 29–31E4, 30–31E5 and 31E6;

- the mean abundances of eggs and larvae from the 1990 surveys are still a reasonable approximation of the current situation;
- the assumptions about the distribution and abundance of ichthyoplankton within the Trevose spawning area will be the same as that within the 'pool', i.e. uniform distribution and abundance throughout the water column.

Within the period February-June, the predicted numbers of eggs and larvae of sea bass entrained by a NNB at Hinkley Point are predicted to be <0.45% of the mean abundance within the Trevose spawning ground. For sole and sprat the numbers of entrained eggs and larvae over the same period are predicted to be <0.005% of the mean abundance within the Trevose spawning ground. However, this entrainment prediction should be used with caution because several assumptions have been made in the estimation and also because interannual variability (which is likely to be great) could not be taken into account. Although we have assumed 100% mortality of all entrained organisms, previous EMU studies have indicated that this may not be the case (but see caveats above), in which case the impacts of entrainment mortality on local populations would be reduced further.

For certain species of conservation interest, such as shads (twaite and Allis) and lampreys (marine and river), that spawn and live as larvae in the freshwater tributaries of the Severn Estuary, entrainment of these early life history stages at Hinkley is expected to be negligible.

5. Predictions and impacts of impingement by species

Crustaceans

The coastal area (out to 6 nautical miles) off the north Devon coast and off the South Wales coast west of the River Rhymney come under the jurisdiction of the Devon and the South Wales Sea Fisheries Committees (SFC), respectively. The sea area of the Bristol Channel east of the Devon and Somerset border around to the mouth of the River Rhymney in South Wales falls outside the geographic boundaries covered by any SFC and, consequently, is an area where fishing activity remains largely unknown. It is suspected that there may be some artisanal crustacean fisheries, for example stake-netting or push-netting for brown shrimps, because healthy populations are known to exist, but the absence of any fisheries authority in the area suggests that it is of relatively little importance from a fisheries perspective. The South Wales SFC suggest that there is little or no potting activity east of Porthcawl on the Welsh coast, and Devon SFC are similarly unaware of any significant potting or trawling activity east of their border.

The official reported landings of shellfish as recorded by the Marine Management Organisation (MMO, previously the MFA) shows that there were no reported landings of brown or pink shrimps from this area in recent years (from 2000). The same data since 2005 show that reported annual landings of brown crab from the Bristol Channel area (as defined by ICES rectangles 30E5, 31E5–E7 and 32E5) are typically of the order of 200 t, but less than 11 t (in 2007) was taken in rectangle 31E6, the eastern portion of which is in the area adjacent to the Somerset coast and in the vicinity of Hinkley Point power station. The level of spatial resolution described by an ICES rectangle prevents us from specifying whether these crabs were taken close to the power station or, more likely, in the extreme west of the area off the North Devon coast. Reported annual landings of velvet swimming crabs (*Necora puber*) and common prawns from the Bristol Channel as a whole since 2005 are of the order of 3.5 t and <200 kg respectively, with just 30 kg of velvet swimming crabs (in 2009 only), and no common prawns coming from rectangle 31E6. Most of the landings of these crustaceans in the Bristol Channel area are made into Devon and Cornwall, or to Welsh ports on the Pembrokeshire coast.

In a national context, the reported landings of these crustaceans into England and Wales in 2008 were: brown crabs, 11 403 t; velvet swimming crabs, 332 t; common prawns, 33 t; brown shrimps, 861 t; shore crabs, 21 t; and pink shrimps, 13 t.

Finfish

The following 13 species constitute about 88% (by number) of the total numbers of finfish impinged at Hinkley B.

Sprat

There is little information on sprat in the Bristol Channel. In the western English Channel the Lyme Bay sprat fishery has been sampled regularly for age and length since 1966, and data show that the back-calculated length at age 1 for sprat in the western Channel is consistently different from that in other regions, despite considerable interannual variation. Although eggs, larvae, juvenile and adult sprat are distributed almost continuously throughout the English Channel, the sprat landed from the western Channel have a markedly different age distribution from those in the east, and it is possible that there is only a limited interchange of sprat from the east and west Channel, and that two stocks exist.

Conclusions

It seems likely that the sprat encountered at Hinkley Point are part of a population that is limited to the Bristol Channel and, given the lack of any assessment for the species, we consider that the most useful comparison for sprat is between impingement data at Hinkley Point power station and landings data reported for UK vessels fishing in the Bristol Channel; ICES statistical rectangles 32 E5–E7, 31 E5–E7 and 30 E5 (sprat = 190 kg).

Based on the scaled-up CIMP dataset, the total annual estimated impingement of sprat at a new power station at Hinkley Point, assuming a constant abstraction rate of 125 cumecs, would be about 3.38 million fish (Appendices B2 and B3). Owing to a lack of biological and population data, it is not possible to derive an EAV for sprat, but as adult sprat are comparatively small, we have assumed an equivalent adult value of unity, although this is likely to be a conservative assumption. With the current cooling water intake design, the equivalent adult numbers of sprat likely to be impinged annually at Hinkley C is ~26.4 t. As the catch of sprat in the local fishery is small (0.19 t currently), this impingement is almost 140 times that of the local fishery. As no stock assessment is made for sprat, it is not possible to assess the impact of impingement on local populations. With the AFD/low-velocity cooling water intake design, the equivalent adult numbers of sprat impinged annually at Hinkley C could be reduced to ~3.16 t, which is about 17 times the local fishery. Sprat are a delicate-bodied pelagic species, and studies conducted at Sizewell B power station indicate that a FRR system is unlikely to reduce impingement mortality (Seaby, 1994; O'Keeffe and Turnpenny, 2005).

Whiting (BAP)

Although the basic biology of whiting is well known, it has proved difficult to estimate its abundance and to follow the dynamics of the different populations around the UK (ICES, 2007b). Part of the problem may be related to distribution and stock structure, and the extent of mixing between areas. However, it is well established that there has been an overall decline in abundance of whiting to very low levels in many areas (ICES, 2008).

There have been sufficient uncertainties in the data used in exploratory assessments for the Celtic Sea whiting (Divisions VIIe–k) stock that ICES is currently unable to provide estimates of fishing mortality or SSB, although SSB shows a decreasing trend and recent recruitment is low (note that survey results indicate that the 2007 year class may be stronger than the recent average).

Conclusions

It seems likely that the whiting encountered at Hinkley Point power station are part of a population that occupies the Bristol Channel and Celtic Sea, with some limited mixing with whiting in the Irish Sea. However, whiting in the latter area are not included in the ICES assessment, whereas the stock assessment area (Divisions VIIe–k) does include the western English Channel. Given this, and the uncertainties expressed by ICES about the reliability of the VIIe–k whiting assessment (which is not used to give management advice), we consider that the most useful comparison is between impingement data at Hinkley Point power station and landings data reported for UK vessels fishing in ICES statistical rectangles 32 E5–E7, 31 E5–E7 and 30 E5 (= 33.48 t, mean 2004–08). At a population level, an indicative (but not robust) comparison is with the SSB estimate for Divisions VIIe–k, weighted by the ratio of the above landings to total UK landings for VIIe-k as used by ICES (on the assumption that these respectively reflect the abundance at the local and "assessment stock" levels). The average UK landings from this stock from 2004 to 2008 were 529 t, and the average annual SSB is estimated at 25,492 t (corresponding to international landings of 9,240 t as estimated by ICES). Therefore, the estimated "local" SSB = 25492 x (33.48/529) = 1613 t.

Based on the scaled-up CIMP dataset, the total annual estimated impingement of whiting at a new power station at Hinkley Point, assuming a constant abstraction rate of 125 cumecs, would be about 2.1 million fish (Appendices B2 and B3). Using the relationship between total numbers, EAV numbers and EAV weights provided by PISCES 2009 to re-scale the impingement estimates derived from the CIMP data, and with the current cooling water intake design, the equivalent adult numbers of whiting likely to be impinged annually at Hinkley C is 288 078 fish (51.28 t). This equates to ~153% of the local whiting fishery (33.5 t) and 3% of the "local" SSB (1724 t). With the AFD/low-velocity cooling water intake design, the equivalent adult numbers of whiting impinged annually at Hinkley C could be reduced to ~23.0 t, which is about 69% of the local fishery and 1.4% of the local SSB. Whiting are a demersal species, though not a robust one such as cod, and studies conducted at Sizewell B power station indicate that a FRR system could reduce impingement mortality by about 50% (Seaby, 1994).

Sole (BAP)

Sole stocks have shown substantial variations in abundance over the past 50 years, largely as a result of fishing and variability in breeding success (Millner and Whiting, 1996). In the more northern regions, the abundance of sole also fluctuates naturally as a result of severe mortality during very cold winters, such as in 1963. The analytical age-based assessment for the sole stock in the Bristol Channel and Celtic Sea (Divisions VIIf and VIIg) is based on landings, two commercial cpue series and one survey index. There is also a confirmatory short UK Fisheries–Science partnership time-series for this and an adjacent area available to the assessors. The general trends in the estimates of stock numbers, fishing mortality and recruitment have been similar in recent assessments. The stock is currently considered by ICES to be fished sustainably and to have full reproductive capacity (ICES, 2008). SSB in 2008 is estimated to be above B_{pa} (2200 t). The average (2003–2007) total annual international catch in VIIf,g (not including discarding) was 1114 t; UK landings were 263 t; and the SSB estimate was 3240 t.

Conclusions

The sole at Hinkley Point are part of a population that occupies the Bristol Channel and Celtic Sea, with relatively limited mixing with adjacent sole populations. We consider that the most valid comparison for sole is between impingement data at Hinkley Point power station and landings data reported for UK vessels fishing in the Bristol Channel and Celtic Sea (Divisions VIIf and VIIg), and with the SSB estimate for this stock. Comparison with a more locally restricted fishery or population, in ICES statistical rectangles 32 E5–E7, 31 E5–E7 and 30 E5, say, would ignore the extensive mixing of early life stages of sole throughout the Bristol Channel and eastern Celtic Sea.

Based on the scaled-up CIMP dataset, the total annual estimated impingement of sole at a new power station at Hinkley Point, assuming a constant abstraction rate of 125 cumecs, would be 602 776 fish (Appendices B2 and B3). Using the relationship between total numbers, EAV numbers and EAV weights provided by PISCES 2009 to re-scale the impingement estimates derived from the CIMP data, and with the current cooling water intake design, the equivalent adult numbers of sole likely to be impinged annually at Hinkley C is 32 429 fish (7.43 t). This equates to ~3% of the local sole fishery (263 t) and 0.23% of the VIIf,g SSB (3240 t). With the AFD/low-velocity cooling-water intake design, the equivalent adult numbers of sole impinged annually at Hinkley C could be reduced to ~6.24 t, which is about 2.4% of the local fishery and 0.2% of the SSB. Sole are a demersal species and studies conducted at Sizewell B power station indicate that a FRR system could reduce impingement mortality by about 96% (Seaby, 1994), but we have assumed here the more conservative estimate of 80% (O'Keeffe and Turnpenny, 2005).

Cod (BAP)

The assessment for cod in ICES Divisions VIIe–k (Western English Channel, Celtic Sea and Bristol Channel) is based on commercial landings, three surveys and four commercial catch per unit effort (cpue) series. Discard data are not included in the assessment, although a correction for high-grading for the years 2003–2005 in the French fisheries has been made. The main uncertainties in this assessment are partial information available on recent quota-induced changes in discarding, and under-reporting and area misreporting of landings. The results of the 2008 assessment are broadly consistent with those of 2007 in terms of trends in fishing mortality, SSB and recruitment, although there was a change in the perception through an upward revision of the 2005 and 2006 year classes by 74% and 67% respectively, and an upward revision of SSB in 2007 by 14%.

ICES (2008) considers cod in Divisions VIIe–k to be overfished, but currently harvested sustainably. The stock has had a truncated age structure over several decades, and its dynamics have been strongly recruitment-driven, i.e. the stock increased in the past in response to good recruitment and decreased rapidly during times of poor recruitment. Fishing mortality has been very high since the mid-1980s, but has declined since 2002 and is now below F_{pa} (0.68). SSB has been below B_{lim} (6300 t) since 2004, but most recently was estimated to be slightly above B_{lim}. Recruitment since 2002 has been well below the long-term average. The average (2003–2007) total annual international catch in VIIe–k (including a high-grading estimate) was 4175 t; UK landings were 343 t; and the estimated SSB was 5133 t.

Conclusions

The cod encountered at Hinkley Point are part of a population that occupies the Bristol Channel and eastern Celtic Sea, with relatively limited mixing with adjacent cod populations. However, the ICES stock assessment area includes the western English Channel and Irish coastal waters, which appear to have cod populations that are separate from the one in VIIf,g. For cod, the most useful comparison is between impingement data at Hinkley Point power station and landings data reported for UK vessels fishing in ICES statistical rectangles 32 E4–E7, 31 E4–E7, 30 E4–E5 and 29 E4 (= 65.17 t, mean 2004–2008). To give some idea of the impact at a stock level, the EAV can be compared with the SSB estimate for Divisions VIIe–k, weighted by the ratio of the above landings to total UK landings from VIIe–k as used by ICES (343 t). Therefore, the estimated "local" SSB = 5133 x (65.17/343) = 975 t.

Based on the scaled-up CIMP dataset, the total annual estimated impingement of cod at a new power station at Hinkley Point, assuming a constant abstraction rate of 125 cumecs, would be about 371 097 fish (Appendices B2 and B3). Using the relationship between total numbers, EAV numbers and EAV weights provided by PISCES 2009 to re-scale the impingement estimates derived from the CIMP data, and with the current cooling water intake design, the equivalent adult numbers of cod likely to be impinged annually at Hinkley C is 32 063 fish (140.4 t). This equates to ~215% of the local cod fishery (65.2 t) and 14% of the local SSB (975 t). With the AFD/low-velocity cooling water intake design, the equivalent adult numbers of cod impinged annually at Hinkley C could be reduced to ~63.1 t, which is about 97% of the local fishery and 6.48% of the local SSB. Cod are a demersal species and studies conducted at Sizewell B power station indicate that a FRR system could reduce impingement mortality by about 94% (Seaby, 1994) but we have assumed the more conservative estimate of 50% (O'Keeffe and Turnpenny, 2005).

Herring (BAP)

Except where a fishery prosecutes spawning herring (e.g. at Llangwm in Milford Haven), larval surveys are the main tool to locate and assess inshore spawning populations, but insufficient numbers of small larvae have been found to assess the status of these small spawning groups of herring. Only MMO landings statistics from local fisheries are available.

Conclusions

It seems likely that the herring encountered at Hinkley Point are part of a population (or populations) that is limited to the Bristol Channel and adjacent inshore waters and, given the lack of any assessment, we consider that the most useful comparison is between impingement data at Hinkley Point power station and herring landings data reported for UK vessels fishing in ICES statistical rectangles 32 E5–E7, 31 E5–E7 and 30 E4–E5 (119.4 t, mean for 2004–2008).

Based on the scaled-up CIMP dataset, the total annual estimated impingement of herring at a new power station at Hinkley Point, assuming a constant abstraction rate of 125 cumecs, would be about 90 526 fish (Appendices B2 and B3). Using the relationship between total numbers, EAV numbers and EAV weights provided by PISCES 2009 to re-scale the impingement estimates derived from the CIMP data, and with the current cooling water intake design, the equivalent adult numbers of herring likely to be impinged annually at Hinkley C is 44 792 fish (5.64 t). This equates to ~5% of the local herring fishery (119.4 t). As no stock assessment is carried out for herring in the area, it is not possible to assess the impact of impingement on local populations. With the AFD/low-velocity cooling water intake design, the equivalent adult numbers of herring impinged annually at Hinkley Point C could be reduced to ~0.28 t, which is about 0.24% of the local fishery. Herring are a delicate-bodied pelagic species, and studies conducted at Sizewell B power station indicate that a FRR system is unlikely to reduce impingement mortality (Seaby, 1994; O'Keeffe and Turnpenny, 2005).

Plaice (BAP)

ICES (2008) advised that the plaice stock in the Celtic Sea (Divisions VIIf,g) had reduced reproductive capacity and was overfished. SSB peaked in the period 1988–1990, following a series of good year classes, then declined rapidly and, since 2002, has been below or around B_{lim} (1100 t). There have been some very weak year classes since the late 1990s. The average (2003–2007) total annual international catch in VIIf,g (not including discarding) was 461 t; UK landings were 84 t; and the SSB estimate was 952 t.

Conclusions

Plaice encountered at Hinkley Point are part of a population that occupies the Bristol Channel and Celtic Sea, with some limited mixing with plaice in the Irish Sea. However, given that ICES conducts separate assessments for 'stocks' in VIIf,g and VIIa (Irish Sea), we consider that the most useful comparison for plaice is between impingement data at Hinkley Point power station and landings data reported for UK vessels fishing in the Bristol Channel and Celtic Sea (Divisions VIIf and VIIg), and with the SSB estimate for this stock. Comparison with a more locally restricted fishery or population, in ICES statistical rectangles 32 E5–E7, 31 E5–E7 and 30 E5, say, would ignore the extensive mixing of plaice life stages throughout the Bristol Channel and eastern Celtic Sea, and with adjacent plaice populations.

Based on the scaled-up CIMP dataset, the total annual estimated impingement of plaice at a new power station at Hinkley Point, assuming a constant abstraction rate of 125 cumecs, would be about 5383 fish (Appendices B2 and B3). Using the relationship between total numbers, EAV numbers and EAV weights provided by PISCES 2009 to re-scale the impingement estimates derived from the CIMP data, and with the current cooling water intake design, the equivalent adult numbers of plaice likely to be impinged annually at Hinkley C is 493 fish (0.23 t). This equates to ~0.3% of the local plaice fishery (84 t) and 0.02% of the Celtic Sea SSB (952 t). With the AFD/low-velocity cooling water intake design, the equivalent adult numbers of place impinged annually at Hinkley C could be reduced to ~0.19 t, which is about 0.23% of the local fishery and 0.02% of the local SSB. Plaice are a demersal species and studies conducted at Sizewell B power station indicate that a FRR system could reduce impingement mortality by up to 100% (Seaby, 1994), but we have assumed the more conservative estimate of 80% (O'Keeffe and Turnpenny, 2005).

Blue whiting (BAP)

The ICES assessment of the stock status of blue whiting is based on an analysis of catch-at-age data from commercial fisheries from 1981 to 2009, and three acoustic surveys that between them cover the distributional area of the spawning stock (ICES, 2010). These show that recruitment of the 2005-2009 year classes has been low (following 10 years of above-average recruitment) and there has been a significant decrease in SSB since 2004, although the estimated abundances for recent years have changed greatly with successive annual assessments. For example, the SSB estimate for 2009 is estimated in 2010 to be about 42% lower than the estimate made in 2009. On the assumption that successive assessments provide retrospective improvements, the ICES (2010) assessment values will be used here.

Conclusions

There is no evidence that blue whiting in the Bristol Channel and Celtic Sea are discrete from the population that occupies the whole of the west coast of northwest Europe (including the Norwegian Sea), which ICES treats as a single stock for assessment purposes. We consider that the most useful comparison is between impingement data at Hinkley Point power station and landings data reported for all vessels fishing the combined stock in Subareas VIII and IX, and Divisions VIId-k (the " southern areas") (= 37 900 t, mean 2004–0008). At a population level, the mean SSB estimate for the whole stock in the years 2004-2008 was 5 360 000 t, which is near the long-term mean for the stock.

Based on the scaled-up CIMP dataset, the total annual estimated impingement of blue whiting at a new power station at Hinkley Point, assuming a constant abstraction rate of 125 cumecs, would be about 1166 fish (Appendices B2 and B3). Using the relationship between total numbers, EAV numbers and EAV weights for whiting (which we have assumed will be similar for blue whiting) provided by PISCES 2009 to re-scale the impingement estimates derived from the CIMP data, and with the current cooling water intake design, the equivalent adult numbers of blue whiting likely to be impinged annually at Hinkley C is 160 fish (0.02 t). With the AFD/low-velocity cooling water intake design, the equivalent adult numbers of place impinged annually at Hinkley C could be reduced to 72 fish (0.01 t). This equates to <0.1% of the blue whiting fishery (37 900 t) and <0.1% of the corresponding SSB (5 360 000 t). Because of the lack of suitable biological data, it has not been possible to assess the impact of the low-velocity cooling water intake design. However, given the low levels of impingement and the large SSB, this would appear to be of minor importance. Assuming that the effectiveness of a FRR for blue whiting is similar to that for whiting (above), a FRR system could reduce impingement mortality by up to 50%.

Specifically designated conservation species:

Eel (Eel management plan)

The EA monitors fish populations extensively within the Severn River Basin District (RBD), although the (mostly) multispecies electric fishing surveys used may underestimate the true density of eel (Knights *et al*., 2001). The data suggest that eels are currently well distributed throughout the lower and middle parts of the catchments, and the EA has concluded that the eel population in the Severn downstream from Worcester has shown little change since the early 1980s, over the period when average recruitment to Europe has declined substantially (by 95% or more; Walker *et al*., 2009). The density and the biomass of eel in the middle reaches of the Severn and Warwickshire Avon catchments were low during the 1980s, but have not been surveyed in recent years. Similar survey data for the Bristol Avon catchment and Somerset rivers within the Severn RBD indicate a general decline in densities and biomasses between 1991 and 1993, and 1994 and 2006, by 37% and 48%, respectively.

A modelling approach to estimate the proportional impact of estuarine glass eel fisheries on the population is available (see Briand *et al*., 2003; Beaulaton and Briand, 2007) and, though it could be used here, it requires extensive sampling of glass eels during spring, when they enter the estuary.

In the absence of data on historical production of eel in England and Wales, a standard production rate of 16.9 kg per hectare has been applied by the Environment Agency in estimating historic production and hence setting the 40% escapement biomass target (6.76 kg per hectare) required under the European Eel Regulation 110/2007. This production rate was selected with reference to estimated production rates for the Bann (Northern Ireland) and Loire (France) catchments, reported by ICES (2008). Using the Environment Agency's Probability Model (see: http://www.defra.gov.uk/foodfarm/fisheries/freshwater/eelmp.htm), silver eel output from the Severn RBD is estimated to be about 8.4 kg per hectare, which equates to about 133.4 t of silver eel per year (Severn Eel Management Plan, March 2010). As such, the Severn RBD is tentatively assessed as exceeding its management target for silver eel production at this time. Note, however, that this model estimate is based on estimates of local yellow eel densities for 109 sites in the Severn catchment, extrapolated to the entire wetted area and converted to silver eel equivalents using a "silvering index", and therefore has a high degree of uncertainty.

The declared annual catches of yellow eels in the years 2005–2008 were 4088, 2785, 892 and 27 kg, respectively, and 419, 968, 134 and 17 kg of silver eels. These annual decreases do not necessarily reflect just changes in eel abundance, but are likely to be attributable too to fluctuations in the fishing effort. Given the small size of the yellow and silver eel fisheries in the Severn RBD, it is not particularly useful to compare these statistics with Her Majesty's Revenue & Customs (HMRC) net export data for eels from the UK as a whole (the best estimate of the UK fishery's catches), and the perceived impact of the Hinkley Point power station can only be evaluated in comparison with the catches declared by the local fisheries.

Currently, eel fishing is banned in the Severn Estuary. However, given that the assumed wetted area is 15881 ha (i.e. 133 400 kg / 8.4 kg ha^{-1}), the 40% escapement biomass target equates to 15 881 x 6.76 = 107.36 t. This leaves a fishery potential of 26 t (i.e. 133.4 – 107.36) if fishing is allowed to resume.

Conclusions

Given Hinkley Point power station's location on the south coast of the Severn Estuary seawards of the River Parrett, the potentially susceptible population consists of glass eels/elvers migrating upstream to freshwater, silver eels migrating downstream from freshwater, and any yellow eels living in the marine environment of the local area. Comparisons of glass eel and yellow/silver eel mortalities through impingement with population estimates are theoretically possible, but the models to permit this are still being developed and it is uncertain anyway which are the relevant 'populations'. The European eel is currently considered to comprise a single reproductive stock throughout its distribution range (and spawns in the Sargasso Sea off the Gulf of Mexico), and individual river and adjacent coastal marine populations appear to mix considerably.

We consider that the most useful indicator of impact is a comparison between impingement data for eels (although these are not differentiated by life stage) at Hinkley Point power station and estimates of the reported catch of each life stage 2005–2008 in the Severn Estuary RBD. A total of 774 kg of glass eels was declared as caught in the Severn RBD in 2005, 684 kg in 2006 and 1254 kg in 2007. The declared annual catches of yellow eels in the years 2005–2007 were 4088, 2785 and 892 kg respectively, and 419, 968 and 133 kg of silver eels.

Based on the scaled-up CIMP dataset, the total annual estimated impingement of eels at a new power station at Hinkley Point, assuming a constant abstraction rate of 125 cumecs, would be about 1304 fish (Appendices B2 and B3) equivalent to 0.08 t of adult eels. As it is not currently possible to derive an EAV for eels because of their complex life history, we have not rescaled the impingement estimates derived from the CIMP data. Nonetheless, with the current cooling water intake design, the equivalent adult numbers of eels likely to be impinged annually at Hinkley C (i.e. 0.08 t) equates to <0.3% of a potential eel fishery (26 t) and <0.06% of the local SSB (133.4 t). Eels are highly unlikely to benefit from lower-velocity cooling water intakes. However, they are considered to be a robust fish and an appropriate FRR system could reduce impingement mortality by up to 100% (Travade and Bordet, 1982), but we have assumed the more conservative estimate of 80% (O'Keeffe and Turnpenny, 2005).

Twaite shad (SAC designated)

Spawning populations of twaite shad are confined to four rivers in the UK, namely the Rivers Tywi, Usk, Wye and Severn (including its tributary the River Teme). The twaite shad is a protected species, but there is only sparse population data for them in the Severn Estuary, so the potential for the estimation of shad stock sizes from current sampling techniques is limited and, as such, few estimates have been made. However, as part of the Severn Tidal Power Feasibility Study Strategic Environmental Assessment, APEM Ltd have recently attempted to estimate shad population size and age distribution using a simplified age-structured matrix model (APEM, 2010). The model applies a matrix incorporating life-history parameters (adult survival rates; sex ratio; fecundity at weight/age; spawning propensity and density-dependence) to predict the number of adult female shad within the River Severn RBD. The model incorporates a density-dependent egg deposition function based on a stock–recruitment relationship derived by M. Aprahamian (pers. comm., cited in APEM, 2010) for adult females aged 6 years and applies forecasting and hindcasting methods using documented life history parameters to predict adult population size in a given year. For the purposes of this study, adults are considered to be aged between 3 and 9 years old.

The model estimate indicates an average population size of approximately 92 000 female shad. Given a sex ratio of 1:1, the total mean population of twaite shad aged between 3 and 9 years in the Severn RBD is therefore estimated to be 184 000, although variation in year-class strength may result in estimates ranging between 112 000 and 596 000.

Based on the scaled-up CIMP dataset, the total annual estimated impingement of twaite shad at a new power station at Hinkley Point, assuming a constant abstraction rate of 125 cumecs, would be about 2276 fish (Appendices B2 and B3). As it is not currently possible to derive an EAV for twaite shad because of the absence of the necessary life history data, we have not rescaled the impingement estimates derived from the CIMP data. Therefore, with the current cooling water intake design, the equivalent adult numbers of twaite shad likely to be impinged annually at Hinkley C (2276 fish) equates to ~1.24% of the estimated local twaite shad population (184 000 adults). With the AFD/low-velocity cooling water intake design, the equivalent adult numbers of twaite shad impinged annually at Hinkley C could be reduced to ~273 fish, about 0.15% of the estimated local population. Twaite shad are a delicate-bodied species, similar to herring and sprat, so we anticipate that a FRR system is unlikely to reduce impingement mortality markedly (O'Keeffe and Turnpenny, 2005).

Allis shad (SAC designated)

Allis shad are rare in the UK. A breeding population of Allis shad was recorded in the River Wye until the 1940s, but individuals are only occasionally found in the Severn Estuary, though they are reported in most years from the River Wye. Given the rarity of the species, population models have not been developed for this species, but for twaite shad alone.

Based on the scaled-up CIMP dataset, the total annual estimated impingement of Allis shad at a new power station at Hinkley Point, assuming a constant abstraction rate of 125 cumecs, would be about 68 fish (Appendices B2 and B3). Allis shad are a delicate-bodied species, similar to herring and sprat, so we anticipate that a FRR system is unlikely to reduce impingement mortality markedly. Given the lack of biological information, it is not possible to derive any estimate for the Allis shad population in the vicinity of Hinkley with which to compare this estimate for impingement.

Lamprey (SAC designated)

More than half the UK SAC designations for the presence of either one or both of river and sea lamprey are situated on the Welsh coast, including the Rivers Wye and Usk. The most recent condition assessment round in 2007 classified all but the River Usk as unfavourable for river lamprey and all but the River Wye as unfavourable for sea lamprey. Stock status information is restricted to SAC rivers and is primarily in the form of ammocoete densities and distribution. The River Usk has the greatest *Lampetra* spp. ammocoete population across all British SAC rivers, and the River Wye has the greatest sea lamprey ammocoete population (APEM, 2007). Although river and sea lamprey are believed to spawn and reside within the River Severn, no assessment has been undertaken of their stock. However, as part of the Severn Tidal Power Feasibility Study Strategic Environmental Assessment, APEM Ltd recently attempted to estimate lamprey population size and age distributions (APEM 2010) using measurements of life-history traits collated from the literature to construct a generic life table for sea lamprey and river lamprey. Lampreys were assumed to represent one discrete population, given the species' capacity to disperse as evidenced by their lack of homing and wide juvenile movement within several rivers throughout the UK. The life cycle of lamprey was represented by a stage-structured model and constructed with vital rate data and information on: average age at metamorphosis (ammocoete and parasitic juvenile); average ammocoete density per m^2 of optimal and suboptimal habitat; metmorphosis success (ammocoete to parasitic juvenile); ammocoete survival; and sex ratio.

Markov Chain Monte Carlo (MCMC) simulations were used to estimate the mean population size from the model output and provide a likely average population size of adult lamprey in the Rivers Usk and Wye. These estimates have been based on best guesses of available habitat of 1% per metre length of river for both optimal and suboptimal habitat. The population estimates are (mean ± s.d.) (APEM, 2010):

River lamprey		**Sea lamprey**	
Usk:	27 667 ± 4696	Usk:	3069 ± 455
Wye:	88 442 ± 14 326	Wye:	12 200 ± 1836
Total:	**116 109**	**Total:**	**15 269**

Based on the scaled-up CIMP dataset, the total annual estimated impingement of river and sea lamprey at a new power station at Hinkley Point, assuming a constant abstraction rate of 125 cumecs, would be about 82 and 207 fish (Appendices B2 and B3), respectively. As it is not currently possible to derive an EAV for lamprey because of their complex life history, we have not rescaled the impingement estimates derived from the CIMP data. Therefore, with the current cooling water intake design, the numbers of lamprey likely to be impinged annually at Hinkley Point C equate to <0.07% of the river lamprey population and 1.36% of the estimated sea lamprey population. Like other similar weakly swimming species, lampreys are unlikely to benefit from AFD/low-velocity cooling water intakes. However, lampreys are considered to be a robust fish and an appropriate FRR system could reduce impingement mortality by up to 100% (Travade and Bordet, 1982), though we have assumed the more conservative estimate of 80% (O'Keeffe and Turnpenny, 2005).

Salmon (SAC designated)

Although estimates of the upstream run of adult salmon are obtained using electronic fish counters or upstream traps on a number of catchments in England and Wales, there are no such data available for rivers entering the Severn Estuary. However, estimates of spawning escapement (numbers of spawning adult fish) are obtained from catch data and exploitation rates, and these are used to assess individual river stock status against conservation limits (CLs: the minimum spawning stock level below which further reductions in spawning numbers are likely to result in significant reductions in the number of juvenile fish produced in the next generation). The CL for each river is defined in terms of eggs deposited.

The River Severn CL is 12.85 million eggs, and the egg deposition estimated for 2008 was 16.56 million, 120% of the CL (mean 131%, 2004–2008). The River Wye CL is 35.66 million eggs, and the egg deposition estimated for 2008 was 22.58 million, 63% of the CL (mean 61%, 2004–2008). The River Usk CL is 10.11 million eggs, and the egg deposition estimated for 2008 was 21.36 million, 211% of the CL (mean 189%, 2004–2008). From these values we can estimate the number of smolts produced, using average egg-to-smolt survival data.

The mean annual catch (2004–2008) of salmon from the Severn Estuary net fishery was 837 fish (the long-term average is ~3000 fish), with rods taking an average of 336, 682 and 987 fish from the Rivers Severn, Wye and Usk, respectively.

Conclusion

For the purposes of evaluating the impact of impingement of salmon smolts or adult fish on the intakes at Hinkley Point power station, data on catches or estimates of abundance for the Severn Estuary and its major rivers, the Severn, Wye and Usk, cover the overwhelming majority of salmon that might be vulnerable. Over the five-year period 2004–2008, the mean annual catch of salmon from the commercial net fishery in the Severn Estuary was 837 fish, and recreational anglers caught an average of 2005 salmon from the Rivers Severn, Wye and Usk combined. Although some 55% of salmon reported caught by anglers on these rivers were released alive, any impact of power station mortalities should be compared with the total catch (not fish killed), because recreational fisheries are valued per salmon caught.

The PISCES 2009 prediction of the total annual estimated impingement at a new power station at Hinkley Point, assuming a constant abstraction rate of 125 cumecs, would be about 276 salmon (see TR065). However, no salmon were recorded in the long-term impingement monitoring programme at Hinkley Point between 2005 and 2009 (see TR065) and none were recorded in the CIMP (Appendix A). We have therefore not made a prediction for salmon in this report.

6. Impacts of impingement and entrainment on the local fish community

In an attempt to assess any long-term impacts of entrainment and impingement during the operation of Hinkley Point B on the local fish community we have analysed the long-term impingement monitoring programme dataset for Hinkley Point collected and collated by Pisces Conservation Ltd (Henderson & Holmes, 1990), see Appendix C.

The 13 most abundant species detected during 1985-89 and in 2004-08 are listed in Table 6 and account for 95.8 % of all the fish impinged in both periods. Of the top 13 species detected from 1985 to 1989, all but two (cod and dab) were also in the top 13 in the years 2004-2008. Both these species were nonetheless still highly ranked (cod 15th and dab 16th) in the latter period.

Table 6 The 13 most abundant species detected in the routine long-term impingement monitoring programme during the periods 1985-1989 and in 2004-2008.

Top 13 species (and numbers) present in 1985-89, in rank order.		**Top 13 species (and numbers) present in 2004-08, in rank order.**	
Whiting	(6397)	Whiting	(7391)
Sprat	(4783)	Sprat	(6687)
Sea snail,Common	(867)	Goby,Sand	(2290)
Goby,Sand	(837)	Sole (Dover sole)	(1576)
Flounder	(413)	Herring	(888)
Pout	(366)	Poor cod	(878)
Sole (Dover sole)	(329)	Pipefish,Snake	(609)
Poor cod	(313)	Rockling,5-Bearded	(462)
Dab	(277)	Flounder	(352)
Bass	(237)	Gurnard,Grey	(349)
Rockling,5-Bearded	(127)	Pout	(346)
Herring	(116)	Sea snail,Common	(320)
Cod	(115)	Bass	(314)

Of the 54 species detected in 2004-08, 15 had not been detected in the years 1985-1989, although none of these were detected in large quantities (Table 7). Similarly, of the 54 species detected between 1985 and 1989, 10 were not detected between 2004 and 2008, and again none of these were detected in large quantities (Table 7).

Table 7 Differences in the presence/absence of fish species detected in the routine long-term impingement monitoring programme between the periods 1985-1989 and 2004-2008.

Species present (and numbers) in 2004-08 but not 85-89			**Species present (and numbers) in 1985-89 but not 2004-08**	
Goby,Painted	(44)		Hake	(51)
Pearlsides	(24)		Goby,Crystal	(3)
Goby,Common	(9)		Anchovy	(3)
Blue Whiting	(5)		Sea bream,Black	(2)
Perch	(2)		Dory (John dory)	(2)
Stickleback,15-spined	(2)		Blenny,Tompot	(1)
Scaldfish	(1)		Wrasse,Cuckoo	(1)
Solenette	(1)		Ling	(1)
Rock cook	(1)		Saithe	(1)
Wrasse,Goldsinny	(1)		Tadpolefish	(1)
Lamprey,River	(1)			
Lamprey, marine	(1)			
Turbot	(1)			
Ray,Blonde	(1)			
Piper	(1)			

An analysis of abundance for 81 species (Appendix C) from 1981 to 2008 shows significant trends for 21 species, of which 15 are positive (i.e. increasing) and 6 are decreasing (eel, dab, sea snail, hake, tadpole fish and Norway pout; note that only 4 tadpole fish were caught over the entire period).

An analysis of abundance trends by species group (Appendix C, see Table 8 for the species composition of each species group) from 1981 to 2008 shows significant trends for 5 of the 10 groups, 3 of which are positive and 2 are negative indicating that, over this 27-year period (Appendix C and Fig. 1). Where a trend is apparent, this would not appear to be a direct result of impingement or entrainment at Hinkley Point B. The decrease in abundance of the “eels, shads, lampreys and salmon” group is essentially driven by the wider decline of those stocks (particularly eels) across the wider Northeast Atlantic in relation to other factors, which is well documented (Dekker, 2008), while the marked increase in the “other pelagic fish” group is largely driven by the a recent population increase in snake pipefish, as observed over the wider Northeast Atlantic during the same period (Harris et al., 2007).

Table 8 The composition of the species groups used in the analysis of abundance trends from 1981 to 2008.

Bass, mullet and sand smelt	Eels, shads, lampreys & salmon	Flatfish	Gadiformes (commercial)
Bass	Twaite shad	Brill	Cod
Golden grey mullet	Eel	Dab	Hake
Sand smelt	River lamprey	Flounder	Ling
Thick-lipped grey mullet	Salmon	Lemon sole	Pollack
Thin-lipped grey-mullet	Sea lamprey	Plaice	Saithe
		Sole	Whiting
Clupeiformes		Turbot	
Anchovy		Witch	
Herring		Scaldfish	
Pilchard		Solenette	
Sprat		Topknot	
Gadiformes (non-commercial)	**Other demersal**	**Other pelagic**	**Scorpaeniformes**
3-Bearded rockling	Blonde ray	Common sandeel	Grey gurnard
5-Bearded rockling	Lesser spotted dogfish	Greater sandeel	Lumpsucker
Blue Whiting	Small eyed ray	Raitt's sandeel	Piper gurnard
Northern rockling	Thornback ray	Garfish	Pogge
Norway pout	Angler fish	Pearlsides	Sea snail,Common
Poor cod	Black seabream	Scad	Tub gurnard
Pout	Conger	Snake pipefish	
Tadpolefish	John dory		
	Red mullet		
Gobiidae	Ballan wrasse		
Black goby	Corkwing wrasse		
Common goby	Cuckoo wrasse		
Crystal goby	Dragonet		
Painted goby	Goldsinny		
Rock goby	Greater pipefish		
Sand goby	Lesser weever		
Transparent goby	Nilsson's pipefish		
	Rock cook		
	Stickleback,15-spined		
	Stickleback,3-Spined		
	Tompot blenny		
	Trigger Fish		
	Worm pipefish		

Figure 1. Abundance trends by species group and year at Hinkley Point B from 1981 to 2008 derived from the long-term impingement monitoring programme dataset (Pisces Conservation Ltd). Annual abundance (vertical axis) is expressed as: ln(1+mean monthly catch).

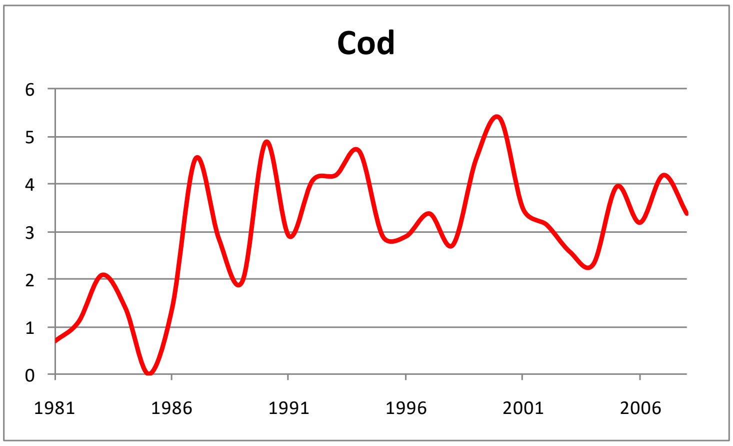

Figure 1 *continued*. Abundance trend for cod at Hinkley Point B from 1981 to 2008 derived from the long-term impingement monitoring programme dataset (Pisces Conservation Ltd). Annual abundance (vertical axis) is expressed as: ln(1+mean monthly catch).

7. Overall Conclusions

7.1 Predicted effects of HP C based upon HP B monitoring data

- An analysis of the abundance trends by species group from 1981 to 2008 from the long-term impingement monitoring programme dataset for Hinkley Point collected and collated by Pisces Conservation Ltd shows that HP B has not had any marked positive or negative effect on the fish community structure at Hinkley Point. Where the data do indicate some population changes (e.g. for European Eel) these would appear to have been the consequence of a change in species abundance across the northeast Atlantic broadly rather than losses to impingement or entrainment at Hinkley Point B itself.
- The proposed HP C has been specified with AFD/low velocity intake structures and a Fish Recovery and Return system. If these proposed impingement mitigation measures function as designed, the impingement losses at HP C are calculated to be similar to, or less than, those of the existing HP B station. The resulting HP C impingement losses will have a negligible effect on the spawning stock of the protected migratory species that use the Severn Estuary and have been captured on the intake screens of HP B (European eel (*Anguilla anguilla*), sea lamprey (*Lampetra fluviatalis*) and twaite shad (*Alosa fallax*). Both the shads are pelagic species, and although limited biological and stock information is available from the literature, we know that they, as all pelagic species, are subject to massive fluctuations in recruitment. These annual fluctuations are related, generally, to environmental stimuli, either directly or indirectly, and can, for relatively small stocks, be as much as an order of magnitude. Given that level of annual variability, our considered

opinion is that an annual 0.5% loss in twaite shad stock as a result of the operation of HPC is not significant and would not have an adverse affect upon the integrity of the SAC based on our scientific understanding of the dynamic nature of such pelagic stocks. The catches of Allis shad and Salmon on the HP B input screens are too small to allow a reliable impingement loss to be calculated. The impact on the commercially important fish species that represent the majority of the existing impingement losses (Sprat, Whiting, Sole, Plaice, Herring and Blue Whiting) is considered to be negligible. For Whiting, Sole, Plaice and Blue Whiting the impingement losses will have a negligible effect on the spawning stock. Sprat is the dominant (>97%) clupeiform fish impinged at HP B and the population trend for this group since 1981 (Figure 1) has remained stable. As HP C (with mitigation) will impinge about half that of the current HP B, we conclude that HP C (with mitigation) is unlikely have any significant impact on local sprat population. For Herring the impingement losses are less than 0.2% of the local fishery and will therefore have negligible impact on the local population. The impact on cod would represent 3.24% of the local SSB. This level of loss is equivalent to 1% of the Total Allowable Catch of cod recommended by ICES for 2011 for Divisions VIIe-k (3420 t) and is unlikely to have any detectable effect on the local cod population when considered against the background natural variability in SSB. The predicted losses of cod from a mitigated HP C are 17% less than those currently caused by HP B. HP B has had no measurable effect on the local abundance trend for cod since 1981.The predicted impingement losses on crustaceans (as represented by the impact on the brown shrimp *Crangon crangon* the main crustacean impinged) are also expected to be similar to those of HP B.

- The community of fish eggs and larvae at Hinkley point is small in both species and numbers and the predicted entrainment losses are insignificant for those species for which we have been able to make an assessment. The impact of entrainment on the shads and lampreys that spawn and live as larvae in the freshwater tributaries of the Severn is expected to be negligible.

7.2 Predicted combined effects of HP C+ HP B

There is a possibility that EDF may apply to extend the operation of the HP B station beyond the current planned closure date of 2016. The remainder of this section considers a notional life extension to 2021. Current construction plans for HP C envisage a staged operation with the first rector operational in 2018, followed by a second rector in 2020. In such circumstances the combined impingement and entrainment effects of HP B + HP C would be:

- 2010 – 2017: Current HP B impact
- 2018-2020: Approximately 1.5 times that of HP B
- 2020 -2021: Approximately 2 times that of HP B

- 2021 on: Approximately the same impact as the existing HP B.

There would therefore be a short period between 2018 and 2021 when the combined impingement and entrainment impact could be up to double that of the existing HP B station, assuming that HP C operates at full capacity immediately. Given the predictions of impingement and entrainment losses from both stations, such a combined effect is expected to remain low with the exception of the possible impact on cod which could rise to approximately 7% of the local SSB during the period 2020-2021, with both stations fully operational. We note too that the term “local” used here is a very large area extending well away from Hinkley Point, so with cod currently under a recovery plan, the current ICES advice being to reduce fishing effort and this stock being among the smallest of the cod stocks of the NE Atlantic, an increase in power-station induced mortality of such a consumer-targeted fish species from the current 3.3% to 7% under a combined HP B + HP C scenario would be noted and may not be considered acceptable by some interested parties.

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Appendix A. Estimated total annual Impingement for HP B and predicted total annual Impingement for HP C by species.

Appendix A: Estimated total annual impingement for Hinkley Point B (A) and predicted total annual impingement for Hinkley Point C (B) by species derived from the BEEMS Comprehensive Impingement Monitoring Programme and Hinkley, 2009-2010.							
	A. Hinkley Point B: Raw data scaled up to total annual impingement, and estimated (bootstrapped) Mean and S.D., assuming an abstraction rate of 33.7 cumecs.				**B. Hinkley Point C: Predictions based on scaled-up data from A with estimated (bootstrapped) Mean and S.D., assuming an abstraction rate of 125 cumecs.**		
Species: common name	**Raw data**	**Mean**	**S.D.**		**Raw data**	**Mean**	**S.D.**
Sprat	946,349	936,386	269,807		3,510,196	3,380,850	1,003,549
Whiting	586,409	578,490	74,455		2,175,110	2,102,759	249,473
Sole, Dover	161,758	159,085	44,479		599,993	602,776	134,369
Cod	103,249	101,075	29,043		382,971	371,097	123,593
Flounder	61,566	63,284	8,245		228,362	219,753	22,685
Mullet, Thin-lipped grey	56,934	56,594	17,300		211,178	216,258	59,174
Rockling, 5-Bearded	39,020	38,650	4,081		144,734	144,019	13,031
Herring	26,282	25,404	6,969		97,486	90,526	23,824
Goby, Sand	20,983	20,198	6,759		77,829	74,724	17,932
Pipefish, Snake	13,277	13,215	3,732		49,247	49,386	12,386
Bass	9,047	9,043	914		33,558	33,562	4,352
Sea snail, Common	8,133	7,562	2,151		30,167	27,262	6,784
Poor cod	2,983	3,029	599		11,063	10,772	2,507
Pout	2,265	2,211	486		8,401	8,433	1,670
Dogfish, Lesser spotted	1,497	1,380	350		5,551	5,262	1,773
Conger	1,480	1,454	231		5,489	5,445	701
Plaice	1,421	1,411	379		5,269	5,383	1,313
Dab	991	972	225		3,676	3,502	888
Pearlsides	920	989	328		3,412	3,570	903
Ray, Thornback (Roker)	877	874	233		3,252	3,325	1,066
Hooknose (Pogge)	852	826	170		3,159	3,090	694
Shad, Twaite	618	646	161		2,292	2,276	694
Rockling, Northern	492	488	171		1,826	1,725	500
Eel	347	351	88		1,288	1,304	270
Whiting, Blue	324	338	196		1,200	1,166	529
Goby, Painted	316	309	146		1,174	1,170	474
Stickleback, 3-Spined	316	326	123		1,171	1,141	473
Lumpsucker	291	278	179		1,078	1,205	530
Sand eel, Greater	267	258	175		990	1,024	493
Turbot	267	297	179		989	838	513
Mullet, Red	266	247	173		986	1,020	495
Pipefish, Nillson's	247	277	123		918	903	491
Pout, Norway	222	209	82		822	794	240
Dragonet	199	197	76		739	690	344
Gurnard, Tub	185	183	81		685	715	256
Smelt, Sand	169	158	40		625	726	150

Appendix A *(continued)*: Estimated total annual impingement for Hinkley Point B (A) and predicted total annual impingement for Hinkley Point C (B) by species derived from the BEEMS Comprehensive Impingement Monitoring Programme and Hinkley, 2009-2010.

	A. Hinkley Point B: Raw data scaled up to total annual impingement, and estimated (bootstrapped) Mean and S.D., assuming an abstraction rate of 30 cumecs.				B. Hinkley Point C: Predictions based on scaled-up data from A with estimated (bootstrapped) Mean and S.D., assuming an abstraction rate of 125 cumecs.		
Species: common name	**Raw data**	**Mean**	**S.D.**		**Raw data**	**Mean**	**S.D.**
Gurnard, Grey	169	170	65		625	587	155
Pollack	150	149	67		558	520	149
Goby, Transparent	112	116	44		415	449	150
Hake	112	126	52		414	346	136
Angler fish	96	97	32		357	347	145
Stickleback, 15-spined	80	89	64		297	283	173
Pipefish, Greater	79	84	52		292	319	166
Wrasse, Ballan	75	70	48		279	288	129
Weever, Lesser	62	74	39		231	210	107
Blenny, Tompot	55	56	25		204	194	121
Lamprey, Sea	52	42	22		193	207	102
Trigger Fish	51	47	37		190	209	138
Rockling, 3-Bearded	42	46	34		155	130	79
Pilchard	31	38	32		117	97	82
Boar fish	23	25	19		86	95	61
Perch	21	20	16		77	77	81
Anchovy	21	17	18		76	66	59
Dory (John dory)	21	26	15		76	79	50
Lamprey, River	21	18	13		76	82	46
Shad, Allis	21	22	15		76	68	40
Rockling, Shore	20	27	23		76	78	61
Brill	19	23	16		71	47	63
Goby, Black	19	23	16		71	47	63
Sand eel, Common	14	13	11		51	56	55
Zander	14	14	13		51	52	52
Dab, Long rough	10	12	11		38	34	40
Gilthead bream	10	11	9		38	43	33
Chub	10	11	10		38	39	35
Shrimp, Grey	5,053,079	4,911,592	899,002		18,742,881	19,135,756	2,958,082
Shrimp, Ghost	3,120,540	3,213,195	503,216		11,574,705	11,448,037	2,294,213
Shrimp, Pink	895,273	896,492	154,938		3,320,747	3,253,744	537,161
Prawn, Atlantic	344,317	348,361	30,633		1,277,141	1,286,515	144,998
Crab, Swimming	12,927	13,004	2,461		47,947	47,988	7,883
Crab, Edible	12,141	11,994	1,645		45,035	45,962	4,537
Crab, Shore	5,258	5,535	740		19,504	18,513	3,406
Jellyfish	3,004	3,162	2,207		11,141	10,751	6,883

Appendix A *(continued):* Estimated total annual impingement for Hinkley Point B (A) and predicted total annual impingement for Hinkley Point C (B) by species derived from the BEEMS Comprehensive Impingement Monitoring Programme and Hinkley, 2009-2010.							
	A. Hinkley Point B: Raw data scaled up to total annual impingement, and estimated (bootstrapped) Mean and S.D., assuming an abstraction rate of 30 cumecs.				**B. Hinkley Point C: Predictions based on scaled-up data from A with estimated (bootstrapped) Mean and S.D., assuming an abstraction rate of 125 cumecs.**		
Species: common name	**Raw data**	**Mean**	**S.D.**		**Raw data**	**Mean**	**S.D.**
Crab, Hermit	2,604	2,577	619		9,658	9,714	1,745
Crab, Velvet swimming	400	377	122		1,482	1,408	421
Cuttlefish, Little	246	256	153		913	832	500
Cuttlefish, European common	82	83	61		306	319	215
Crab, Hairy	79	69	41		295	320	161
Crab, Long-legged spider	75	75	35		279	265	94
Crab, Sardine	41	42	33		152	157	97
Lobster	20	27	23		76	78	61
Krill	10	11	10		38	34	40

Appendix B1. Estimated total annual impingement for HP B for selected species assuming an abstraction rate of 30 cumecs via current intake structures compared with local fishery and estimated local population size.

Appendix B2. Predicted total annual impingement for HP C for selected species assuming an abstraction rate of 125 cumecs via current intake structures compared with local fishery and estimated local population size.

Appendix B3. Predicted total annual impingement for HP C for selected species assuming an abstraction rate of 125 cumecs via current and low velocity intake structures compared with local fishery and estimated local population size.

Appendix B4. Predicted total annual impingement for HP C for selected species assuming an abstraction rate of 125 cumecs via current and low velocity intake structures and with a fish recovery and return system compared with local fishery and estimated local population size.

Appendix B1: Estimated total annual impingement at Hinkley Point B for selected species assuming an abstraction rate of 33.7 cumecs via current intake structures compared with local fishery and estimated local polulation size. ("NA" indicates no assessment made)							
Species: common name	**Number**	**EAV (number)**	**EAV (wt, t)**	**local fishery (t)**	**local SSB (t)**	**%of local fishery**	**% local SSB**
Sprat (largest numbers)	936,386	936,386	7.30	0.19	NA	3844%	-
Whiting (BAP)	578,490	79,253	14.11	33.5	1613	42%	0.87%
Sole (BAP)	159,085	8,559	1.96	263	3240	1%	0.06%
Cod (BAP)	101,075	8,733	38.25	65.2	975	59%	3.92%
Herring (BAP)	25,404	12,570	1.58	119.4	NA	1%	-
Plaice (BAP)	1,411	129	0.06	84	952	0%	0.01%
Blue whiting (BAP)	338	46	0.00	37,900	5,360,000	0%	0.00%
Eel (Eel management plan)	351	351	0.02	-	133.4	-	0.02%
Twaite shad (SAC designated)	646	646		-	184,000	-	0.35%
Allis shad (SAC designated)	22	22	-	-	-	-	-
Sea lamprey (SAC designated)	42	42	-	-	15,269	-	0.27%
River lamprey (SAC designated)	18	18	-	-	116,109	-	0.02%
Salmon (SAC designated)	0	0	-	-	-	-	-
Crangon crangon (the main crustacean impinged)	4,911,592						
Notes: EAV estimates for whiting, sole, cod, herring and plaice have been derived from PISCES 2009, the EAV for blue whiting has been derived using data for whiting from PISCES 2009. The EAVs for sprat and both species of shad has been assumed as 1.							

Appendix B2: Predictied total annual impingement at Hinkley point C for selected species assuming an abstraction rate of 125 cumecs via current intake structures compared with local fishery and estimated local polulation size. ("NA" indicates no assessment made)

Species: common name	Number	EAV (number)	EAV (wt, t)	local fishery (t)	local SSB (t)	%of local fishery	% local SSB
Sprat (largest numbers)	3,380,850	3,380,850	26.37	0.19	NA	13879.3%	-
Whiting (BAP)	2,102,759	288,078	51.28	33.5	1613	153.1%	3.18%
Sole (BAP)	602,776	32,429	7.43	263	3240	2.8%	0.23%
Cod (BAP)	371,097	32,063	140.44	65.2	975	215.4%	14.40%
Herring (BAP)	90,526	44,792	5.64	119.4	NA	4.7%	-
Plaice (BAP)	5,383	493	0.23	84	952	0.3%	0.02%
Blue whiting (BAP)	1,166	160	0.02	37,900	5,360,000	0.0%	0.00%
Eel (Eel management plan)	1,304	1,304	0.08	-	133	-	0.06%
Twaite shad (SAC designated)	2,276	2,276	-	-	184,000	-	1.24%
Allis shad (SAC designated)	68	68	-	-		-	-
Sea lamprey (SAC designated)	207	207	-		15,269	-	1.36%
River lamprey (SAC designated)	82	82	-		116,109	-	0.07%
Salmon (SAC designated)	0	0	-	-	-	-	-
Crangon crangon (the main crustacean impinged)	19,135,756						

Notes: EAV estimates for whiting, sole, cod, herring and plaice have been derived from PISCES 2009, the EAV for blue whiting has been derived using data for whiting from PISCES 2009. The EAVs for sprat and both species of shad has been assumed as 1.

Appendix B3: Predictied total annual impingement at Hinkley point C for selected species assuming an abstraction rate of 125 cumecs via low-velocity intakes and with an acoustic fish deterrent (AFD) system compared with local fishery and estimated local polulation size. ("NA" indicates no assessment made)									
Species: common name	**Number (current intake)**	**EAV (number, current intake)**	**estimated entrapment risk**	**EAV Number (with AFD)**	**EAV (wt, t) with AFD)**	**local fishery (t)**	**local SSB (t or number)**	**% of local fishery**	**% of local SSB**
Sprat (largest numbers)	3,380,850	3,380,850	0.12	405,702	3.16	0.19	NA	1666%	-
Whiting (BAP)	2,102,759	288,078	0.45	129,635	23.08	33.5	1613	68.88%	1.43%
Sole (BAP)	602,776	32,429	0.84	27,241	6.24	263	3240	2.37%	0.19%
Cod (BAP)	371,097	32,063	0.45	14,428	63.20	65.2	975	96.93%	6.48%
Herring (BAP)	90,526	44,792	0.05	2,240	0.28	119.4	NA	0.24%	-
Plaice (BAP)	5,383	493	0.84	414	0.19	84	952	0.23%	0.02%
Blue whiting (BAP)	1,166	160	0.45	72	0.01	37,900	5,360,000	0.00%	0.00%
Eel (Eel management plan)	1,304	1,304	1	1,304	0.08	-	133.4		0.30%
Twaite shad (SAC designated)	2,276	2,276	0.12	273		-	184,000		0.15%
Allis shad (SAC designated)	68	68	0.12	8		-			-
Sea lamprey (SAC designated)	207	207	1	207		-	15,269		1.36%
River lamprey (SAC designated)	82	82	1	82		-	116,109		0.07%
Salmon (SAC designated)	0	0	-	-	-	-	-	-	-
Crangon crangon (the main crustacean impi	19,135,756	19,135,756	1						
Notes: EAV estimates for whiting, sole,cod, herring and plaice have been derived from PISCES 2009, the EAV for blue whiting has been derived using data for whiting from PISCES 2009. The EAVs for sprat and both species of shad has been assumed as 1. The estimated reduction in entrapment risk is based on Maes et al. 2004 and the value for blue whiting has been assumed to be the same as that for whiting.									

Appendix B4: Predictied total annual impingement at Hinkley point C for selected species assuming an abstraction rate of 125 cumecs via current intake structures with an acoustic fish deterrent (AFD) system and with a fish recovery and return (FRR) system compared with local fishery and estimated local polulation size. ("NA" indicates no assessment made)

Species: common name	Number (current intake)	EAV (number, current intake)	estimated entrapment risk	estimated recovery through FRR	EAV Number (AFD & FRR)	EAV (wt, t) AFD & FRR)	local fishery (t)	local SSB (t or number)	AFD & FRR: %of local fishery	AFD & FRR: % local SSB
Sprat (largest numbers)	3,380,850	3,380,850	0.12	0%	405,702	3.16	0.19	NA	1665.5%	-
Whiting (BAP)	2,102,759	288,078	0.45	50%	64,818	11.54	33.5	1613	34.4%	0.72%
Sole (BAP)	602,776	32,429	0.84	80%	5,448	1.25	263	3240	0.5%	0.04%
Cod (BAP)	371,097	32,063	0.45	50%	7,214	31.60	65.2	975	48.5%	3.24%
Herring (BAP)	90,526	44,792	0.05	0%	2,240	0.28	119.4	NA	0.2%	-
Plaice (BAP)	5,383	493	0.84	80%	83	0.04	84	952	0.0%	0.00%
Blue whiting (BAP)	1,166	160	0.45	50%	36	0.00	37,900	5,360,000	0.0%	0.00%
Eel (Eel management plan)	1,304	1,304	1	80%	261	0.08	-	133.4	-	0.06%
Twaite shad (SAC designated)	2,276	2,276	0.12	0%	273		-	184,000	-	0.15%
Allis shad (SAC designated)	68	68	0.12	0%	8		-		-	
Sea lamprey (SAC designated)	207	207	1	80%	41		-	15,269	-	0.27%
River lamprey (SAC designated)	82	82	1	80%	16		-	116,109	-	0.01%
Salmon (SAC designated)	0	0	-	-	-	-	-	-	-	-
Crangon crangon (the main crustacean impinged)	19,135,756	19,135,756		80%						

Notes: EAV estimates for whiting, sole,cod, herring and plaice have been derived from PISCES 2009, the EAV for blue whiting has been derived using data for whiting from PISCES 2009. The EAVs for sprat and both species of shad has been assumed as 1. The estimated reduction in entrapment risk is based on Maes et al. 2004 and the value for blue whiting has been assumed to be the same as that for whiting.

Appendix C Trend Analysis of the long term HP B fish Impingement data

The long term HP B impingement monitoring data from 1981 to 2008 has been subject to statistical analysis to determine whether there are any significant trends in the data on the basis of individual species, ecological guild or the species group.

Functional Guild Classification after Elliot and Hemingway, 2002

Code	Description
ER	Estuarine resident taxa
FW	Freshwater taxa
CA	Diadromous taxa
MS	Marine seasonal taxa
MJ	Marine juvenile taxa
MA	Marine adventitious taxa

One simple way to look at trend over a period of years is to use the Mann-Kendall statistic (Mann, 1945; Kendall, 1975). For a particular series (a species, guild or group), this looks at all pairs of counts (Cj,Ck) such that j>k. If Ck>Cj then the pair scores a 1, if Ck<Cj then the pair scores a -1, if they are the same then the score is 0. The statistic MK is the sum of all these scores. Thus, an increasing series would have a positive score; a decreasing series would have a negative score. To make comparisons easier, the statistic has been standardised by dividing by the number of paired comparisons. Thus, if there was a perfect increasing series then the statistic would have value +1; if there was a perfect decreasing series then the statistic would have value -1.

This statistic only measures trend in some average sense over the whole range of years. Thus, it could detect generally increasing positive or negative trends. The statistic will not be able to tease out more subtle situations where, for example, the trend increases and then decreases. Thus, it is important to consider this statistic in conjunction with plots of the data.

For a more formal definition of the MK statistic, consider the following. Denote the set of n (n=27 for our series) observations by Y_j (j=1,…,n) and the set of m observations which occur in a later year than Y_j by Y_k (k=1,…,m). We then calculate the statistic

$$MK = \sum_{j=1}^{n}\sum_{k=1}^{m} I(Y_j, Y_k) / n(n-1)$$

where I is an indicator variable defined by the sign of D = $Y_j - Y_k$. If D is positive then I = 1, if D is negative then I = -1, if D = 0 then I = 0.

Under the alternative hypothesis of some form of trend (not specifying positive or negative trend) we can calculate p-values for our observed value of the absolute value of MK using Monte-Carlo simulation. This is done by simulating the MK statistic 1000 times (each time with the n data points randomly re-ordered) and then calculating the absolute value of MK. This null distribution is used to calculate p-values for the observed value of the absolute value of MK by calculating the proportion of the null distribution that is greater than our observed value. All computations were carried out in R (R Development Core team, 2010).

Results

The MK statistic and its associated p-value have been calculated for each of the 81 species, for the 5 guilds and the 10 groups. For the guilds and groups the counts for species making up those guilds and groups have been summed. The results are shown in the table below with rows highlighted for which the p-value was less than 0.05 (taken as the level to define statistical significance).

There are a number of species with statistically significant trends; the direction of the trend is given by the sign of the MK statistic.

Only one of the guilds has a statistically significant trend (MJ). The guild trends are plotted in Figure 1 and suggest that MJ has the biggest trend.

Five of the groups have statistically significant trends; these are plotted as group trends in Figure 2. For 'other pelagic' the plot has been done twice: once with all of the data, and once removing a large value (486) in 2008. This allows the increasing trend in later years to be seen more easily.

Species	MK	p-value
[1,] Agonus cataphractus (L.)	0.0199	0.77
[2,] Alosa fallax (Lacepede)	-0.1083	0.113
[3,] Ammodytes marinus	0.0142	0.6235
[4,] Ammodytes tobianus L	0.037	0.5495
[5,] Anguilla anguilla (L.)	-0.3632	0.0005
[6,] Aphia minuta (Risso)	-0.0214	0.7505
[7,] Atherina boyeri Risso	0.1097	0.0725
[8,] Balistes carolinenis (Gmelin)	0.037	0.236
[9,] Belone bellone (L.)	0.0085	0.776
[10,] Blennius gattorugine L.	-0.0114	0.814
[11,] Buglossidium luteum (Risso)	0.0499	0.1115
[12,] Callionymus lyra L.	0.0598	0.384
[13,] Centrolabrus exoletus (L.)	0.0071	0.792
[14,] Ciliata mustela (L.)	0.2265	0.0015
[15,] Ciliata septentrionalis (Collet)	0.1382	0.034

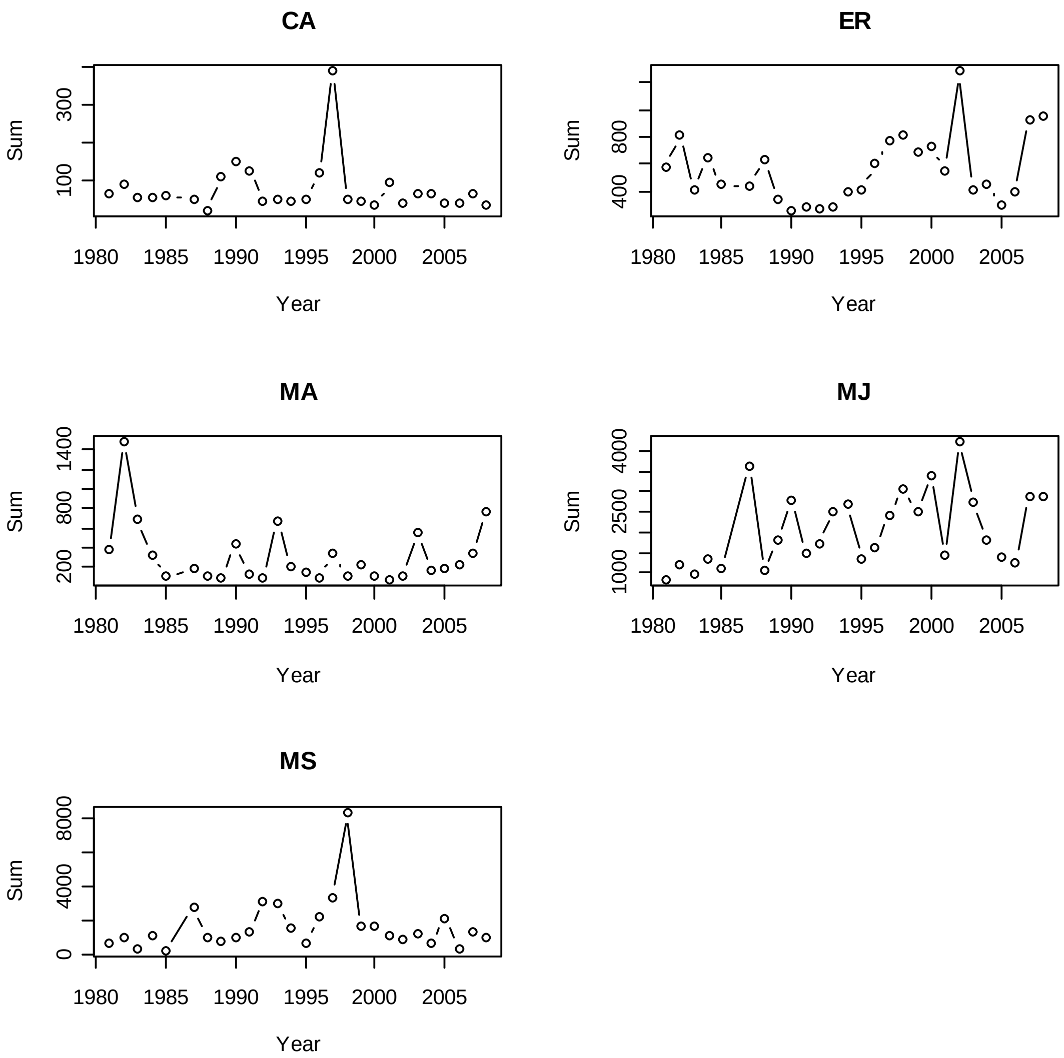
[16,] Clupea harengus L.	0.2892	0.0005
[17,] Conger conger L.	-0.0057	0.924
[18,] Crenilabrus melops (L.)	0.0028	0.917
[19,] Crenimugil labrosus (Risso)	0.0185	0.682
[20,] Crystallogobius linearis	-0.0641	0.1555
[21,] Ctenolabrus rupestris (L.)	-0.057	0.223
[22,] Cyclopterus lumpus L.	-0.1182	0.0735
[23,] Dicentrarchus labrax (L.)	0.1467	0.0295
[24,] Engraulis encrasicolus (L.)	-0.0527	0.1615
[25,] Entelurus aequoreus (L.)	0.1239	0.071
[26,] Eutrigla gurnardus (L.)	0.2308	0.0015
[27,] Gadus morhua L.	0.1439	0.0285
[28,] Gaidropsaurus vulgaris	-0.0028	0.9485
[29,] Gasterosteus aculeatus L.	0.1667	0.0065
[30,] Glyptocephalus cynoglossus	-0.0142	0.629
[31,] Gobius niger L.	0.0741	0.189
[32,] Gobius paganellus	0.0142	0.6365
[33,] Hyperoplus lanceolatus	0.0655	0.142
[34,] Labrus bergylta Ascanius	-0.0299	0.5465
[35,] Labrus mixtus	-0.0114	0.7895
[36,] Lampetra fluviatalis (L.)	0.0114	0.8405
[37,] Limanda limanda (L.)	-0.198	0.0035
[38,] Liparis liparis (L.)	-0.1638	0.0195
[39,] Liza aurita	0.0513	0.2185
[40,] Liza ramada (Risso)	0.1453	0.0405
[41,] Lophius piscatorius L.	-0.0897	0.0815
[42,] Maurolicus muelleri (Gmelin)	0.245	0.0005
[43,] Merlangius merlangus (L.)	0.1353	0.054
[44,] Merluccius merluccius (L)	-0.2778	0.0005
[45,] Micromesistius poutassou	0.1125	0.033
[46,] Microstomus kitt	0.0171	0.6855
[47,] Molva molva (L.)	-0.0256	0.487
[48,] Mullus surmuletus L.	0.104	0.098
[49,] Nerophis lumbriciformis	0.0057	0.864
[50,] Petromyzon marinus	0.0484	0.139
[51,] Platichthys flesus (L.)	0.0954	0.1545
[52,] Pleuronectes platessa L.	0.1752	0.0045
[53,] Pollachius pollachius (L.)	-0.0627	0.3565
[54,] Pollachius virens (L.)	-0.0199	0.467
[55,] Pomatoschistus microps	-0.0299	0.6095

[56,] Pomatoschistus minutes	0.1353	0.059
[57,] Pomatoschistus pictus	0.1667	0.0065
[58,] Psetta maxima (L.)	0.0128	0.8155
[59,] Raja brachyura	0.0199	0.521
[60,] Raja clavata L.	-0.0114	0.8675
[61,] Raja microocellata	-0.0142	0.6355
[62,] Raniceps raninus (L.)	-0.0912	0.0265
[63,] Salmo salar L.	-0.0299	0.589
[64,] Sardina pilchardus (Walbaum)	-0.0128	0.6825
[65,] Scophthalmus rhombus (L.)	-0.0641	0.242
[66,] Scyliorhinus caniculus (L.)	0.1254	0.049
[67,] Solea solea L.	0.2977	0.0005
[68,] Spinachia spinachia (L.)	0.0043	0.933
[69,] Spondyliosoma cantharus (L.)	-0.0855	0.1235
[70,] Sprattus sprattus (L.)	0.0328	0.6215
[71,] Syngnathus acus (L.)	0.1595	0.011
[72,] Syngnathus rostellatus Nillson	-0.0527	0.446
[73,] Trachinus vipera Cuvier	-0.0057	0.847
[74,] Trachurus trachurus (L.)	-0.0399	0.5105
[75,] Trigla lucerna L.	-0.0883	0.156
[76,] Trigla lyra	0.0185	0.5715
[77,] Trisopterus esmarkii	-0.1396	0.036
[78,] Trisopterus luscus (L.)	0.0043	0.9535
[79,] Trisopterus minutus (L.)	-0.0527	0.456
[80,] Zeugopterus punctatus (Bloch)	0	0.9855
[81,] Zeus faber L.	-0.0427	0.183

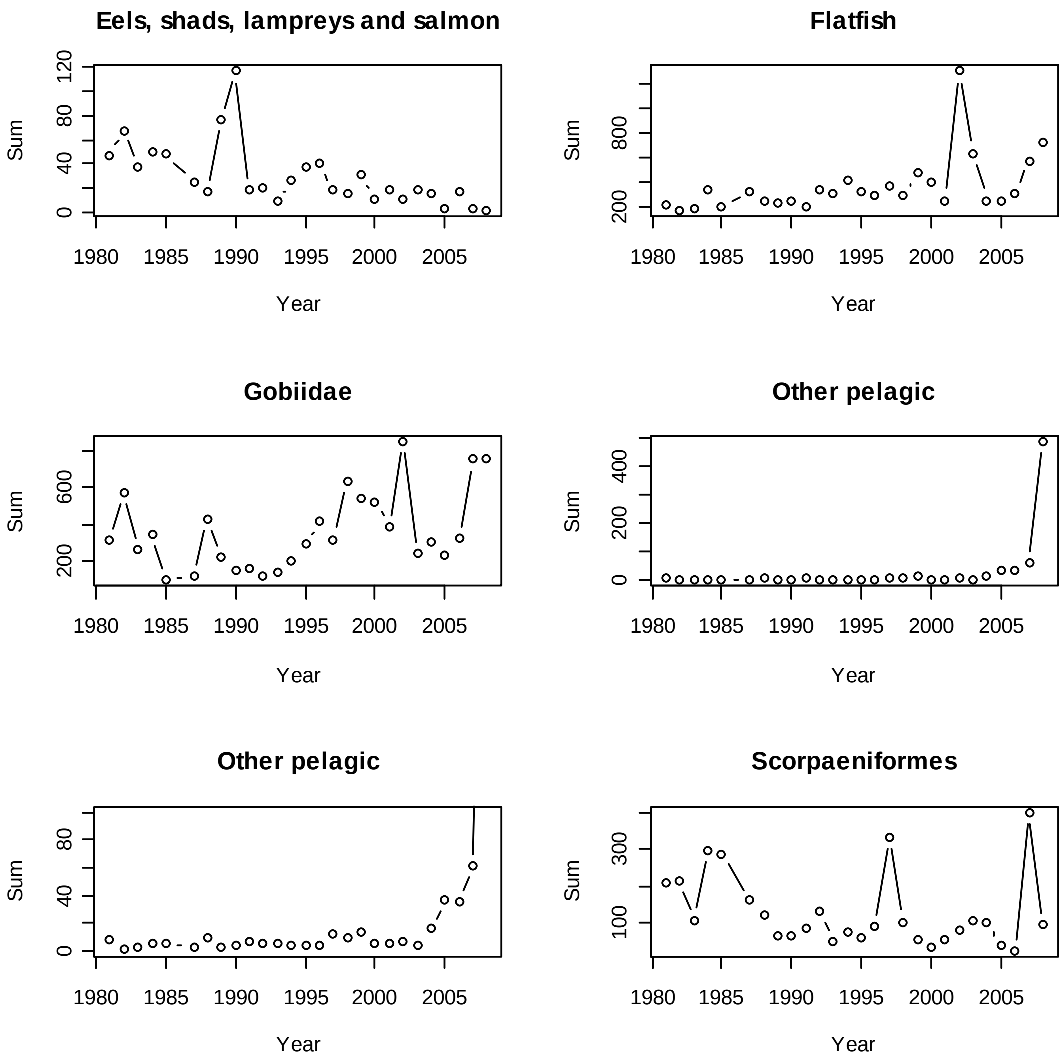
Guild	**MK**	**p-value**
CA	-0.114	0.114
ER	0.0741	0.2985
MA	-0.0085	0.898
MJ	0.1781	0.0045
MS	0.0556	0.389

Group	**MK**	**p-value**
Bass, mullet and sand smelt	0.1054	0.1235
Clupeiformes	0.0556	0.422
Eels, shads, lampreys and salmon	-0.2778	0.0005
Flatfish	0.2208	0.0005

Gadiformes (commercial)	0.1325	0.054
Gadiformes (non-commercial)	-0.0242	0.731
Gobiidae	0.1524	0.03
Other demersal	0.0299	0.681
Other pelagic	0.2422	0.0005
Scorpaeniformes	-0.1524	0.0195



Guild plots



Group Plots for where p-value<0.05

The Sizewell C Project

6.3 Volume 2 Main Development Site

Chapter 22 Marine Ecology and Fisheries

Appendix 22I - Sizewell C Impingement Predictions Based Upon Specific Cooling Water System Design

Revision: 1.0

Applicable Regulation: Regulation 5(2)(a)

PINS Reference Number: EN010012

May 2020

Planning Act 2008
Infrastructure Planning (Applications: Prescribed Forms and Procedure) Regulations 2009

Sizewell C – Impingement predictions based upon specific cooling water system design

Sizewell C – Impingement predictions based upon specific cooling water system design

Table of contents

Executive summary **10**

1.1 Revisions to impingement assessments 14

1.1.1 V2 report dated 9/12/2019 14

1.1.2 V3 report dated 17/01/2020 14

1.1.3 V4 report dated 28/01/2020 15

1.1.4 V5 report dated 19/02/2020 15

1.1.5 V6 report dated 27/02/2020 15

2 Background **17**

2.1 Selection of key taxa for SZC impingement assessment 17

2.1.1 Conservation species impingement data 18

2.2 Aims of this report 19

3 Selection of impingement mitigation measures for SZC **20**

3.1 Biofouling and other blockage risks 20

3.2 Impingement Mitigation Optioneering 21

3.3 Biota exclusion technology 21

3.3.1 Passive wedge-wire cylinder screens (PWWC screens) 22

3.3.2 Bubble curtains 23

3.3.3 Behavioural deterrents – strobe lights 23

3.3.4 Behavioural deterrents - electric barriers 23

3.3.5 Behavioural deterrents - Acoustic fish deterrents (AFDs) 23

3.3.6 Intake design - LVSE intakes 24

3.4 Biota Recovery Technology 25

3.4.1 Fish Recovery and Return System 25

3.5 Conclusions 26

4 Relevant site features **26**

5 Impingement Assessment Methodology **28**

5.1 Introduction 28

5.1.1 Differences in the abundance of fish at the SZB and SZC intake locations 28

5.1.2 Effect of the proposed SZC intake heads upon the expected impingement rate 29

5.2 SZB impingement data collection and collation 29

5.3 Estimated annual mean, minimum and maximum losses by the SZB station 32

5.4 Statistical method used to derive annual impingement estimates at Sizewell 33

5.5 Predicted annual mean, minimum and maximum losses by the SZC station 34

5.6 Effect of the intake head design 34

5.7 FRR system mortality 34

5.7.1 Trash rack mortality 34

5.7.2 FRR survival 37

5.7.3 Other factors which could potentially affect FRR survival rates 37

5.7.4 Values used for FRR survival in the SZC impingement assessment 39

5.8 EAV conversion factors 40

5.8.1 Discussion of the Spawner Production Foregone method for calculating EAVs 41
5.8.2 Potential detail change to the EAV calculation methodology in TR383 41
5.9 Conversion from EAV numbers to equivalent weight.. 42
5.10 Evaluating the effect of SZC impingement losses – comparison with ICES stock estimates 43
5.10.1 Are ICES stock units appropriate for assessing the effects of SZC on fish populations?... 44
5.10.2 Scientific status of the ICES Bass stock unit and the consequent SSB estimate............... 45
5.10.3 Conclusions on the validity of ICES stock units... 46
5.10.4 How does ICES deal with non-fishing impacts on stocks?... 47
5.11 Evaluating the effect of SZC impingement losses – Comparison with international landings data 47
5.12 Evaluating the effect of SZC impingement losses – Other comparative data sources 47
6 Assessment of the significance of impingement effects 51
6.1 Screening thresholds for negligible effects in context... 51
6.1.1 What is meant by a sustainable fish population?.. 51
6.1.2 Natural variability of fish stocks 52
6.1.3 Comparison with sustainable levels of harvesting rate for data rich stocks 53
6.1.4 An example of where screening thresholds for fish mortality have been applied for major infrastructure projects.. 54
6.1.5 The appropriateness of a 1% SSB screening threshold for impingement effects 54
6.1.6 Appropriateness of a 1% SSB threshold for the endangered European eel 55
7 Impingement predictions for SZC - finfish.. 56
7.1 Predicted impingement without embedded mitigation measures 56
7.2 Predicted SZC impingement with LVSE intake heads fitted... 56
7.3 Predicted SZC impingement with FRR systems fitted 56
7.4 Predicted SZC impingement with the effect of the LVSE intake heads and FRR systems fitted . 57
7.5 Further consideration of impingement effects on bass and thin-lipped grey mullet..................... 57
7.6 Consideration of the impingement losses of finfish species of conservation concern 64
7.6.1 Cucumber Smelt 64
7.6.2 River lamprey and sea lamprey 65
7.6.3 Twaite shad and allis shad 66
7.6.4 European eel 69
7.6.5 North Sea Herring and Blackwater Herring 77
7.7 SZC predicted entrapment effects (impingement + entrainment)... 79
7.8 Consideration of potential local effects on the fish assemblage at Sizewell 87
7.8.1 Fish at Sizewell in a southern North Sea context 87
7.8.2 What local effects of SZC entrapment might be important?.. 88
7.8.3 SZC Impingement Risk Zones and the potential for very local fish depletion 89
7.8.4 Evidence of localised impingement effects from other sites.. 89
7.8.5 SZC entrapment effects on smelt in the Alde Ore and potentially in the Blyth water bodies 90
7.8.6 Conclusions on potential local effects from SZC entrapment.. 91
7.9 Contextualising SZC entrapment losses 91
8 Shellfish impingement predictions for SZC... 92
8.1 FRR system mortality.. 92

8.1.1 Trash rack mortality ... 92
8.1.2 FRR survival ... 93
8.2 EAV conversions ... 93
8.3 Evaluating SZC impacts on shellfish ... 94
8.4 Predicted SZC impingement effects on shellfish without embedded mitigation measures ... 94
8.5 Predicted SZC impingement effects with the LVSE intake heads fitted ... 95
8.6 Predicted SZC impingement effects on shellfish with FRR systems fitted ... 95
8.7 Conclusions on the effects of SZC on shellfish ... 95

9 Consideration of climate change effects ... 95

9.1 Changes in the Sizewell fish community ... 96
9.2 Potential future changes ... 97
9.3 Effect on SZC impingement predictions ... 98

10 Effect of SZC entrapment on the Water Framework Directive (WFD) status of local water bodies 99

11 Conclusions ... 100

References ... 102

Appendix A Cooling water system design ... 109

A.1 Main cooling water systems in each pumping station ... 109

Appendix B Calculated annual impingement by number at SZB and SZC without mitigation – all species 112

Appendix C Mean, lower and upper numbers of fish estimated (SZB) and predicted SZC) to be impinged annually – full calculation tables ... 118

C.1 Predicted impingement without embedded mitigation measures ... 118
C.2 Predicted SZC impingement with the effect of LVSE intake heads ... 120
C.3 Predicted SZC impingement with FRR systems fitted ... 122
C.4 Predicted SZC impingement with the effect of LVSE intake heads and FRR systems fitted ... 124

Appendix D Mean numbers of fish estimated impinged annually at SZB ... 128

D.1 Unmitigated impingement effects ... 128
D.2 Impingement effects with embedded FRR mitigation ... 129

Appendix E Number at age and proportion maturity of commonly impinged fish species ... 130

E.1 Sprat ... 130
E.2 Herring ... 130
E.3 Whiting ... 131
E.4 Bass ... 131
E.5 Sole ... 132

List of Tables and Figures

Tables

Table 1 Predicted reduction in impingement mortality for SZC fitted with LVSE intakes and FRR system compared with an unmitigated SZC. 11

Table 2 Annual mean SZC predictions of impingement for the 24 key species with the proposed LVSE intake heads and FRR systems fitted and the corrections to bass and thin lipped grey mullet assessments incorporated as per Section 7.5. Losses have been converted to adult equivalent (EAV) numbers and weights (t) and calculated as a percentage of the mean stock SSB (t) or, if this is not available, mean international landings (t). Species where the impingement weight > 1 % of the relevant stock comparator (given in bold) would be shaded red (there are none). Note, values in red font are estimates of the population numbers (e.g. sand goby) or reported catch numbers (salmon & sea trout) 16

Table 3. Predicted reduction in impingement mortality for SZC fitted with LVSE intakes and FRR system compared. 26

Table 4 Summary of the number of sampling visits to SZB completed between 2009 and 2017 32

Table 5 Proportion of fish, by species that will not pass through the 75 mm wide trash racks, and the length size used for the cut-off 36

Table 6 Predicted FRR mortality by species through the SZC drum and band screens 40

Table 7 EAV metrics and mean weight of individuals used to convert the numbers impinged to adult equivalent numbers and weights at SZC. See BEEMS Technical Report TR383 for full EAV calculations. Species where an EAV could not be calculated are highlighted in yellow and the impingement losses are overestimates 42

Table 8 Data sources used to provide information on relevant stock unit, landings and SSB 49

Table 9 Modelled year to year variations in SZB impingement numbers (2009-2017) 53

Table 10 Sustainable fishing mortality values based upon a precautionary management approach for species relevant to Sizewell 54

Table 11 Annual mean SZC impingement predictions with no impingement mitigation. Losses have been converted to adult equivalent (EAV) numbers and weights (t) and calculated as a % of either the mean stock SSB (t) or mean international landings (t). Species where the impingement weight > 1 % of the relevant stock comparator (either SSB or landings – given in bold) are shaded red. Numbers in red font are estimates of the population numbers (e.g. sand goby, smelt, twaite shad, allis shad) or catch numbers (salmon & sea trout) 59

Table 12 Annual mean SZC impingement predictions considering the effect of the intake head design. Losses have been converted to adult equivalent (EAV) numbers and weights (t) and calculated as a % of either the mean stock SSB (t) or mean international landings (t). Species where the impingement weight > 1 % of the relevant stock comparator (either SSB or landings – given in bold) are shaded red. Numbers in red font are estimates of the population numbers (e.g. sand goby, smelt, twaite shad, allis shad) or reported catch numbers (salmon & sea trout) 60

Table 13 Annual mean SZC impingement predictions with FRR systems fitted (no adjustment for the intake head design). Losses have been converted to adult equivalent (EAV) numbers and weights (t) and calculated as a % of either the mean stock SSB (t) or mean international landings (t). Species where the impingement weight > 1 % of the relevant stock comparator (either SSB or landings – given in bold) are shaded red. Numbers in red font are estimates of the population numbers (e.g. sand goby, smelt, twaite shad, allis shad) or reported catch numbers (salmon & sea trout) 61

Table 14 Annual mean SZC impingement predictions considering the effect of the intake head design and with FRR systems fitted. Losses have been converted to adult equivalent (EAV) numbers and weights (t) and calculated as a % of either the mean stock SSB (t) or mean international landings (t). Species where the impingement weight > 1 % of the relevant stock comparator (either SSB or landings – given in bold) are shaded red. Numbers in red font are either estimates of the population numbers (e.g. sand goby, smelt, twaite shad, allis shad) or reported catch numbers (salmon & sea trout) 62

Table 15 Annual mean SZC impingement predictions considering the effect of the LVSE intake heads and FRR systems fitted and the corrections to thin lipped grey mullet and bass assessment detailed in Section 7.5. Losses have been converted to adult equivalent (EAV) numbers and weights (t) and calculated as a % of either the mean stock SSB (t) or mean international landings (t). Numbers in red font are either estimates of the population numbers (e.g. sand goby, smelt, twaite shad, allis shad) or reported catch numbers (salmon & sea trout) 63

Table 16 Annual mean SZC entrapment predictions (impingement + entrainment) **with no impingement mitigation**. For impingement, losses have been converted to adult equivalent (EAV) numbers and weights (t) and calculated as a % of either the mean stock SSB (t) or mean international landings (t) for the period 2009-2017. For entrainment, the worst-case losses have been converted to EAV numbers and weight and calculated as a % of the SSB and landings in 2010 only. Species where the entrapment weight > 1 % of the relevant stock comparator (either SSB or landings – given in bold) are shaded red. Numbers in red font are either estimates of the population numbers (e.g. sand goby) or reported catch numbers (salmon & sea trout) 81

Table 17 Annual mean SZC entrapment predictions (impingement + entrainment) **considering the effect of the intake head design and with FRR systems fitted**. Losses have been converted to adult equivalent (EAV) numbers and weights (t) and calculated as a % of either the mean stock SSB (t) or mean international landings (t). Species where the impingement weight > 1 % of the relevant stock comparator (either SSB or landings – given in bold) are shaded red. Numbers in red font are either estimates of the population numbers (e.g. sand goby) or reported catch numbers (salmon & sea trout) 83

Table 18 Annual mean SZC entrapment predictions (impingement + entrainment) considering the effect of LVSE intake heads and FRR systems fitted and the corrections to thin lipped grey mullet and bass impingement assessment detailed in Section 7.5 (main changes shown in yellow). Losses have been converted to adult equivalent (EAV) numbers and weights (t) and calculated as a % of either the mean stock SSB (t) or mean international landings (t). Species where the impingement weight > 1 % of the relevant stock comparator (either SSB or landings – given in bold) would be shaded red (there are none in this Table) with the exception of sand goby where a 10% of SSB or landings comparator has been used. Numbers in red font are either estimates of the population numbers (e.g. sand goby) or reported catch numbers (salmon & sea trout) 85

Table 19 Differences in age and maturity of fish caught at Sizewell and Hinkley Point as reflected in Equivalent Adult Values (EAV) where an EAV of 1 is a mature adult. 88

Table 20 SZC equivalent abstraction after impingement mitigation of LVSE intakes and FRR system in comparison with an unmitigated SZC abstracting 132 cumecs 90

Table 21 Comparison of mean fishery landings as a percentage of SSB with predicted SZC mean entrapment as a percentage of SSB for the period 2009-2017. 91

Table 22 Discards by year as a percentage of landed fish weight compared with predicted SZC entrapment as a percentage of landed fish weight. 92

Table 23 Proportion of shellfish, by species that will not pass through the 75 mm wide trash racks 93

Table 24 Proportion mortality by species through the SZC drum and band screens 93
Table 25 EAV metrics and mean weight of individuals used to convert the numbers impinged to adult equivalent numbers and weights of shellfish at SZC. (See BEEMS Technical Report TR383 for brown crab EAV calculations) 93
Table 26 Data sources used to provide information on relevant stock unit, landings and SSB.. 94
Table 27 ICES Rectangles in Southern North Sea CFU... 94
Table 28 ICES rectangles used in East Anglia LFU 94
Table 29 Annual mean SZC impingement predictions with no impingement mitigation for key shellfish species. Losses have been converted to adult equivalent (EAV) numbers and weights (t) and calculated as a % of the mean international landings (t). Species where the impingement weight > 1 % of the landings – given in bold) are shaded red 95
Table 30 Annual mean SZC impingement predictions with FRR mitigation fitted for shellfish species. Losses have been converted to adult equivalent (EAV) numbers and weights (t) and calculated as a % of either the mean international landings (t). Any species where the impingement weight > 1 % of the landings – given in bold) are shaded red 95
Table 31 Annual estimated numbers of fish impinged by SZA in 1981-1982 (Turnpenny and Utting, 1987) and by SZB in 2009-2017 (BEEMS Technical Report TR339), the number at SZA raised to the SZB pumping capacity, the percentage of the total number impinged and the species' rank.. 97
Table 32 WFD Transitional Fish Classification Index metrics 99
Table 33 Cooling water flow volumes when SEC/CFI systems are supplied from the band screens 111
Table 34 Cooling water flow volumes when SEC/CFI systems are supplied from the drum screens 111

Figures

Figure 1 The coast at Sizewell, showing the locations of the intake and outfall for SZB and the proposed intakes and outfalls for SZC. The locations of three intake locations are shown for each SZC tunnel but only 2 heads will be fitted per tunnel with locations dependent upon geotechnical considerations. 27
Figure 2 The SZC CIMP impingement assessment process 31
Figure 3 Comparison between estimated daily impingement weights for ctenophores and fish from SZB CIMP data. 39
Figure 4 ICES stock units for bass (ICES WGCSE 2019) 46
Figure 5 Southern North Sea and Channel Natura 2000 sites designated for Twaite shad 68
Figure 6 EA monitoring data for upriver glass eel migration from the Blackwater in Essex. 72
Figure 7 SZB impinged eel length frequency 2009-2017. The peak at 6.75cm corresponds to the 2 glass eels that were impinged in the period... 75
Figure 8 SZB impinged eel cumulative length frequency distribution 2009-2017. 75
Figure 9 Variation in mean impingement rates by month at SZB... 76
Figure 10 Herring impingement numbers at SZB (CIMP programme) by year 78
Figure 11 Change in the Blackwater herring SSB from 2010 to 2018 (Source Cefas)............... 79
Figure 12 Illustrative schematic of EPR cooling water circuits for each unit (Source EDF CNEPE E.T.DOMA/09 0119 A1 Approved). The equalising pond shown in the figure is the station forebay and SZC has 1 forebay for each unit.. 110

Please note that the red line boundary was amended after this document was finalised, therefore figures in this document do not reflect the boundaries in respect of which development consent has been sought in this application. However, amendments to the red line boundary does not have any impact on the findings set out in this document and all other information remains correct.

Executive summary

EDF Energy plans to build a new coastal nuclear power station (Sizewell C, SZC), adjacent to the operational Sizewell B (SZB) and decommissioned Sizewell A (SZA) sites in Suffolk. The station would be of a once-through design, abstracting large volumes of seawater for cooling the condenser steam. As part of the application for the building and operation of the new station, EDF Energy is required to evaluate the effects that the abstraction of seawater may have on the marine environment. The Centre for Environment, Fisheries and Aquaculture science (Cefas) supported by a network of subcontractors has been contracted by EDF Energy to undertake the necessary marine studies to provide the evidence base for the SZC DCO application via a comprehensive set of studies known collectively as the BEEMS programme for SZC.

SZC would need to abstract approximately 132 cumecs ($m^3\ s^{-1}$) compared with approximately 51.5 cumecs for the existing SZB. SZB is the most recent nuclear power station to be constructed in the UK (commissioned in 1995) and is fitted with two measures to reduce the losses of impinged fish and crustacea; specially designed capped intake head and an early example of a Fish Recovery and Return (FRR) system. The cooling water intakes for SZC would be protected by widely spaced bars to prevent the intake of cetaceans, seals and large items of debris, but a significant number of small organisms (small fish and crustaceans, and plankton) will inevitably enter the cooling water intakes. The larger organisms must be removed before the water enters the power station cooling system to prevent them blocking the condenser tubes. These organisms are removed through impingement on rotating fine-mesh (10 mm at SZB, also proposed for SZC) drum screens which protect the main cooling water supply to the station condensers and band screens that protect the essential and auxiliary cooling water systems. The smaller organisms (mostly fish eggs and larvae and other plankton) that pass through the drum screens are entrained and pass through the power station cooling system without causing significant blockages.

Impinged organisms will be returned to the sea via a Fish Recovery and Return system (FRR). Not all will survive this process and separate assessments have been made to:

1. evaluate the impact of the loss of impinged organisms on fish populations (this report TR406)
2. evaluate the effect of any returned dead fish on local water and ecological quality. (BEEMS Technical Report TR520)
3. determine whether any beached fish would constitute a nuisance on local beaches. (BEEMS Technical Report TR511).

A separate report considers the significance of entrainment impacts on marine organisms (BEEMS Technical Report TR318) but the results are summarised in this report (TR406) to produce a combined entrapment assessment for SZC (Section 7.7).

Ninety-one finfish taxa were recorded at Sizewell over the 9-year study period. Of these 24 species have been selected as being representative of the fish assemblage and which include species of importance commercially, ecologically and from a conservation perspective. Similarly, four shellfish species were selected for assessment on the basis of commercial and ecological importance. Where possible impingement and entrapment predictions and compared against internationally coordinated stock assessments of agreed stock units for each species.

Selection of impingement mitigation technology for Sizewell C

The Environment Agency have issued guidelines for the types of measures that could be adopted at new direct cooled power stations to reduce the predicted environmental impacts of impingement and indirectly the potential for pollution by discharges of dead fish back into the marine environment (Environment Agency 2005, 2010). As explained in the Environment Agency guidelines, in practice the selection of potential impingement mitigation measures involves a complex consideration of the likely effectiveness of each measure in the marine environment at the station location, engineering feasibility and operational safety for staff and the plant. The range of options is much larger in freshwater and some brackish environments that do not present a high biofouling risk to station plant and for low volume abstractions (e.g. of a few cumecs) but many of these options are infeasible for a coastal direct cooled power stations (Environment Agency 2005). SZC's intakes would abstract 132 cumecs and would be mounted on the seabed in a highly turbid, coastal environment with high wave exposure offshore of the Sizewell-Dunwich Bank. The site is at high risk of biofouling.

A detailed consideration of the effectiveness and feasibility of the available impingement mitigation options has been conducted for Sizewell C and is summarised in Section 3. These studies demonstrated that two measures were both feasible and likely to reliably deliver reductions in the predicted losses of fish and crustacea:

i. Low velocity side entry (LVSE) intake heads
ii. Fish Recovery and Return (FRR) system with proposed 10mm mesh filtration and an anti-biofouling control policy that results in chlorination not being applied to the FRR system.

Both of these technologies are proposed SZC station and their use is predicted to reduce impingement mortality at SZC by the factors shown in Table 1 compared with an unmitigated SZC.

Table 1 Predicted reduction in impingement mortality for SZC fitted with LVSE intakes and FRR system compared with an unmitigated SZC.

Group	Example species	Impingement reduction at SZC
Pelagic fish	sprat, herring, anchovy, shads	62%
Demersal fish	bass, cod, whiting, grey mullet	77-79%
Epibenthic fish	eel, lampreys, sole, sand goby	92%
Shellfish	crab, lobster, brown shrimp	92%

Assessment of the significance of SZC impingement effects

There are no formal UK regulatory guidelines for assessing the significance of fish mortality levels caused by impingement in coastal power stations and therefore any assessment must be based on expert judgment.

For the purposes of this assessment we have adopted two screening thresholds that have been selected such that impingement losses lower than the appropriate threshold will have negligible effects on the year to year sustainability of a fish population. Effects above the appropriate threshold would not necessarily indicate a significant adverse effect but require further investigation to determine whether significant effects were, in fact, present.

The thresholds have been selected based upon internationally accepted scientific practice for the sustainability of fish stocks under anthropogenic pressures:

a. For commercially exploited stocks and conservation species (which includes stocks that are not currently exploited): 1% of the Spawning Stock Biomass (SSB) or, as a highly conservative proxy, 1% of international landings of the stock.
b. For unexploited stocks: 10% of the SSB or, as a highly conservative proxy, 10% of international landings of the stock.

The scientific rationale for the selection of these screening thresholds is detailed in Section 6.

For eel, twaite shad, allis shad, cucumber smelt and river lamprey a more precautionary approach was adopted of comparing SZC effects with 1% of a geographically limited subset of the entire stock. In particular, a highly precautionary approach was adopted for European eel whereby the Anglian River Basin District (RBD) SSB was used as the stock reference due the uncertainties surrounding both the current eel stock status and its stock dynamics (Sections 6.1.6, 7.6.4). This is equivalent to adopting a highly precautionary threshold of approximately 0.005% SSB for the eel stock.

Derivation of Spawning Stock Biomass estimates

Fish stocks in the Northeast Atlantic are managed partly through the EU Common Fisheries Policy (CFP), whose objective is to maintain or rebuild fish stocks to levels that can produce their maximum sustainable yield (MSY). The International Council for the Exploration of the Sea (ICES) advises public authorities with competence for marine management including the European Commission (EC).

ICES' advice is produced through a process which is set up to ensure that the advice is based on the best available science and data, is considered legitimate by both authorities and stakeholders and is relevant and operational in relation to the policy in question.

The basis for the advice is the compilation of relevant data and analysis by experts in the field, normally through an expert group which includes core researchers in the field. This analysis is peer reviewed by scientists who have not been involved in the expert group and have no direct interest in the matter.

To support the stock by stock management system, ICES provides advice on fishing opportunities and stock status for individual stocks including estimates of Spawning Stock Biomass (SSB). To undertake their stock assessments ICES' scientists have identified biological stock areas that describe the distribution of a stock. These may be different from the areas defined by the EU, for example, for the management of fishing quotas and technical measures. Identification of appropriate stock boundaries has been a central theme of ICES' coordinated effort since its formation in 1902 and major advances in understanding have, and continue to be, made.

Are ICES stock units appropriate for assessing the effects of SZC on fish populations?

The appropriateness of using some of the existing ICES stock units, particularly for bass which has one of the largest stock units of the key fish species included in the SZC effects assessment needs to be considered. In particular, whether the stock areas being used for the assessment of impacts to certain species consider the impact to local sub-populationsgiven numerous papers (including papers produced by ICES) provide evidence of sub-populations and more complex heterogeneous population structures.

Section 5.10.1 describes how ICES determines stock identity for fisheries management purposes, in particular how it uses evidence from ongoing research. The status of the bass stock unit and the direction of ongoing research is addressed and found not to alter the decision that ICES' current stock definition is scientifically the most appropriate. The section concludes that ICES' stock boundaries are compromises but they are based on a mature weighing of the best scientific evidence available and they are relied upon by governments to manage fish populations in the waters of all EU member states. Given the negligible predicted SZC impacts compared to those of fishing, and the precautionary nature of ICES' estimates of SSBs, no justification is found not to use the ICES' stock definitions to assess SZC effects on fish.

Assessment Results

Predictions of impingement have been provided for SZC without mitigation (Table 11), with mitigation that separately includes LVSE intake heads (Table 12) and FRR systems (Table 13) fitted and for the station fitted with both of the two mitigation technologies (

Table 15).

The individual entrainment and impingement impacts are such that when combined into a single entrapment estimate, there is very little difference to the overall conclusions that are reached when each is viewed separately. In the absence of impingement mitigation, species that exceed the 1 % threshold are bass, thin-lipped grey mullet, European eels and sand gobies. With the proposed impingement mitigation fitted entrapment estimates (Table 18) show the only species that remains above the 1 % threshold is sand goby (entrainment = 1.4 % of abundance; impingement = 0.0 %, i.e. entrapment = 1.4 %). Sand gobies are a short-lived very abundant species that is ubiquitous in European coastal areas to at least a depth of 20 m.

The species produces pelagic larvae which are dispersed by tidal currents resulting in a lack of genetic diversity over the southern North Sea. Given that the species is not commercially-exploited, the appropriate negligible effects threshold is 10% of SSB (as discussed in Section 6.1). However, because of their short lifespan and early age of maturity, sand gobies have a sustainable harvesting rate of greater than 50% SSB (Section 6.1.1). Therefore, losses of 1.4 % of total abundance by SZC are regarded as negligible.

Conclusions on the predicted effects of SZC entrapment

a. Of the 24 key fish and 4 key shellfish species, no species exceeded the 1% impingement screening threshold for negligible effects with the proposed LVSE intake heads and FRR systems fitted when compared against stock estimates or, in the absence of these, international landings. For the European eel, twaite shad, allis shad, cucumber smelt and river lamprey assessments a more precautionary approach was adopted of comparing SZC effects with 1% of a geographically limited subset of the entire stock. For eel this was equivalent to adopting a highly precautionary screening threshold of approximately 0.005% SSB. The predicted impingement effects of SZC on all of the key taxa were negligible (Table 2).

b. The predicted effects of SZC entrapment (i.e. impingement plus entrainment) with the proposed embedded impingement mitigation systems fitted were also negligible (Table 18).

c. An assessment of potential localised effects of SZC entrapment was undertaken (Section 7.8) and found no likely significant adverse effects on:
    i. spawning or nursery areas in the vicinity of Sizewell
    ii. the prey of HRA protected breeding little tern (the potentially most vulnerable species to localised effects on prey fish abundance at Sizewell).

Effect of SZC entrapment on the Water Framework Directive (WFD) assessment of local water bodies Section

Section 10 considers whether SZC entrapment has the potential to cause deterioration in the status of surface water bodies (both within and between status classes) by adversely affecting the fish biological quality element of two nearest transitional water bodies to Sizewell:

i. Blyth (S) at approximately 12 km to the north of Sizewell

ii. Alde & Ore at approximately 25 km to the south of Sizewell

The assessment concluded that SZC entrapment would have no significant effect on the calculated WFD fish biological quality element - the Transitional Fish Classification Index (TFCI). There would, therefore, be no predicted change in the WFD status of the Blyth (S) and Alde & Ore transitional water bodies due to SZC entrapment. This assessment included a specific consideration of the likelihood of any significant effects of SZC entrapment on smelt in the Alde Ore at the request of stakeholders. The conclusion of that study was that no significant effects are expected.

1.1 Revisions to impingement assessments

1.1.1 V2 report dated 9/12/2019

Since Version 1 of this report was released (03/06/2019), new details and clarifications on the proposed station design have emerged. Also, additional work has been undertaken to address issues raised by stakeholders. This has resulted in a number of revisions to this report both minor internal updates and more significant updates issued to stakeholders The major changes/impacts on the assessments that are included in this report are:

1. After a detailed modelling programme EDF Energy have decided to fit low velocity side entry (LVSE) intake heads at SZC. These intakes are designed to substantially reduce impingement impacts. Calculations have been undertaken to assess the effects of the proposed head design on SZC impingement and in conjunction with the proposed FRR mitigation.

2. The fine filtration mesh size is proposed to be 10 mm and the associated trash rack bar spacing proposed 75mm. No adjustments are therefore required for the mesh size as this is consistent with that of the current SZB station. A proposed wider trash rack spacing required an update to the proportion of each species that will either pass through or be retained on the trash racks. Trash rack mortality calculations have been updated accordingly.

3. Improved estimates of the populations of the conservation species; twaite shad, cucumber smelt and river lamprey have been included. This has substantially improved the confidence in impact assessment for these species. All tables have been updated accordingly.

4. Following stakeholder comments, estimates of the sand goby population have been updated. A calculation error was noted in the adjustment for beam trawl sampling efficiency. The population estimate has been re-scaled to the expected abundance if the trawl was fishing at 100 % efficiency.

5. Impingement estimates for key crustacea (brown crab, brown shrimp, lobster and whelks have been included in this version of the report.

6. All Appendices have been updated to the latest values. Following on from a stakeholder request, the final Appendix table for SZC (including all mitigation), now includes all calculation steps, including trash rack mortality.

7. An overall entrapment assessment has been included that presents the combined effects of impingement plus entrainment at SZC (Section 7.7).

1.1.2 V3 report dated 17/01/2020

1. Following the release of Version 2 of this report, an error was spotted with some of the stock data in that catch data (landings plus discards) were used, rather than landings only. This error affected four species only (whiting, cod, horse mackerel and dab). Assessments for the first 3 species were unaffected by the error as the primary stock comparator for these species is SSB and not landings. For dab, the result of replacing catches with landings was negligible – in the absence of mitigation, losses of dab changed from 0.01 % of catches to 0.04 % of landings. All calculations and tables for the four species have been updated.

2. Sections 5.7 on factors that could influence FRR mortality and 5.8 on EAV calculations have been expanded to provide clarifications in response to stakeholder comments. In particular, this report now provides:

    - An assessment of the potential for clogging of the FRR system by dead fish and ctenophores

- A critique of potential alternative methods of calculating EAVs

These were editorial changes only and they resulted in unchanged assessment results.

3. Following discussions with stakeholders at MTF meetings, further clarifications and explanations have been provided on the assessment of the predicted effects of SZC on:

    i. the endangered European eel stock in sections 6.1.6 and 7.6.4

    ii. the North Sea herring and the Blackwater herring in Section 7.6.5.

    iii. the local fish assemblage in Section 7.8.

4. A new Section 7.9 has been added to contextualise predicted SZC entrapment losses. The data included are an expanded version of those provided to stakeholders at the Sizewell MTF meeting on 18 December 2019.

1.1.3 V4 report dated 28/01/2020

1. Clarification of the development site red line boundary information added after the Table of Figures

1.1.4 V5 report dated 19/02/2020

In response to stakeholder comments clarifications have been added about:

1. The measured differences in bass abundance at the SZB and SZC intake locations; in particular how the surveys were intended to quantify previously well-established scientific facts about bass thermal preference in winter. Section 5.1.1.1.

2. Details of impinged eel length frequency distributions from SZB impingement data added in Section 7.6.4.3.

3. In response to Environment Agency comments dated 12 February 2012 on the v2 report. In particular, the following sections have been added:

    - Section 3 to summarise the extensive range of studies and the decision-making process that took place on the selection of impingement mitigation options to be fitted at SZC.

    - Section 5.10 which provides a justification of the use of SSBs derived from ICES's stock units to assess the effects of SZC on fish populations. A specific SZC question on the validity of the bass stock unit size is addressed in Section 5.10.2

    - Section 10 - Effect of SZC entrapment on the Water Framework Directive (WFD) status of local transitional water bodies. This assessment included a specific consideration of the likelihood of any significant effects of SZC entrapment on smelt in the Alde Ore (and potentially the Blyth) at the request of the Environment Agency.

1.1.5 V6 report dated 27/02/2020

1. Minor editorial correction to 5.1.1.1

    Table 9 updated with Sizewell C data to replace previous reference to Hinkley Point data.

Table 2 Annual mean SZC predictions of impingement for the 24 key species **with the proposed LVSE intake heads and FRR systems fitted and the corrections to bass and thin lipped grey mullet assessments incorporated as per Section 7.5**. Losses have been converted to adult equivalent (EAV) numbers and weights (t) and calculated as a percentage of the mean stock SSB (t) or, if this is not available, mean international landings (t). Species where the impingement weight > 1 % of the relevant stock comparator (given in bold) would be shaded red (there are none). Note, values in red font are estimates of the population numbers (e.g. sand goby) or reported catch numbers (salmon & sea trout)

Species	Mean SZC prediction (No mitigation)	SZC prediction with LVSE intakes	FRR mortality	EAV number	EAV weight (t)	mean SSB	% of SSB	Mean landings (t)	% of landings
Sprat	7,125,393	2,729,025	2,729,025	2,050,190	21.53	220,757	**0.01**	151,322	0.01
Herring	2,555,783	978,865	978,865	700,103	132.08	2,198,449	**0.01**	400,244	0.03
Whiting	1,865,492	714,484	393,295	140,044	40.03	151,881	**0.03**	17,570	0.23
Bass	57,537	22,037	12,133	2,717	4.16	14,897	**0.03**	3,051	0.14
Sand goby	381,612	146,157	30,108	30,108	0.06	205,882,353	**0.01**	NA	NA
Sole	250,059	95,773	19,729	4,200	0.90	43,770	**0.00**	12,800	0.01
Dab	148,921	57,037	30,715	13,656	0.56	NA	NA	6,135	**0.01**
Anchovy	73,865	28,290	28,290	27,558	0.57	NA	NA	1,625	**0.04**
Thin-lipped grey mullet	67,684	25,923	14,273	1,190	0.62	600	**0.10**	120	0.52
Flounder	38,180	14,623	3,377	1,559	0.13	NA	NA	2,309	**0.01**
Plaice	25,288	9,685	1,995	689	0.17	690,912	**0.00**	80,367	0.00
Smelt	23,863	9,139	9,139	6,959	0.12	105,733,825	**0.01**	8	1.36
Cod	16,845	6,451	3,884	1,395	3.63	103,025	**0.00**	34,701	0.01
Thornback ray	10,802	4,137	852	164	0.52	NA	NA	1,573	**0.03**
River lamprey	6,720	2,574	530	530	0.04	62	**0.07**	1	3.76
Eel	4,516	1,730	356	356	0.12	79	**0.15**	14	0.84
Twaite shad	3,601	1,379	1,379	1,379	0.43	7,519,986	**0.02**	1	32.40
Horse mackerel	4,077	1,561	1,561	1,561	0.22	NA	NA	20,798	**0.00**
Mackerel	628	241	241	241	0.08	3,888,854	**0.00**	1,026,828	0.00
Tope	64	24	5	5	0.03	NA	NA	498	**0.01**
Sea trout	10	4	4	4	0.01	NA	NA	39,795	**0.01**
Allis shad	5	2	2	2	0.00	27,397	**0.01**	0	0.68
Sea lamprey	5	2	0	0	0.00	NA	NA	NA	NA
Salmon	0	0	0	0	0.00	NA	NA	38,456	**0.00**

2 Background

EDF Energy plans to build a new coastal nuclear power station (Sizewell C, SZC), adjacent to the operational Sizewell B (SZB) and decommissioned Sizewell A (SZA) sites in Suffolk. The station will be of a once-through design, abstracting large volumes of seawater for cooling the condenser steam. As part of the application for the building and operation of the new station, EDF is required to evaluate the effects that the abstraction of water may have on organisms in the marine environment. Although the cooling water intakes will be protected by widely spaced bars to prevent the intake of cetaceans, seals and large items of debris, a significant number of small organisms (small fish and crustaceans, and plankton) will inevitably enter the cooling water intake. The larger organisms must be removed before the water enters the power station cooling system to prevent them blocking the condenser tubes. These organisms are removed through impingement on rotating fine-mesh (10 mm at SZB, also proposed for SZC) drum screens which protect the main cooling water supply to the station condensers and band screens that protect the essential and auxiliary cooling water systems. The smaller organisms (mostly fish eggs and larvae and other plankton) that pass through the drum screens are entrained and pass through the power station cooling system without causing significant blockages. Impinged organisms will be returned to the sea via a Fish Recovery and Return (FRR) system. Not all will survive this process and EDF Energy is required to evaluate the effect of the loss of these organisms on marine communities and also of the potentially polluting effects of dead biota discharged from the FRR system on local water quality and marine ecology.

As was the case for Hinkley Point C (HPC), the impingement assessment process for SZC makes use of extensive impingement data collected at the adjacent power station. However substantial differences between the two existing power stations have created very different assessment datasets and necessitated a more complex statistical modelling based approach for the SZC assessments (Section 5). At HPC, assessment of impingement losses was made using data collected at the Hinkley Point B (HPB) station (BEEMS Technical Report TR456). The assessment used data collected in 2009 – 2010 from a sampling programme, known as the Cefas Comprehensive Impingement Monitoring Programme (CIMP), which comprised 40 * 24-h samples to estimate annual impingement of the station. The predictions were supported using information from the lower-resolution Routine Impingement Sampling Programme, which has sampled for over 35 years at the HPB site, but at a lower sampling effort.

Prior to the BEEMS programme, there was no regular impingement sampling at the SZB site. To fill this data gap, a CIMP programme was initiated in 2009 to provide the necessary data for predictions of impingement for the proposed SZC site. The CIMP was designed to provide 24-h sampling of the fish, invertebrates and other material passing through the SZB cooling water systems on 28 to 40 occasions per year. Between February 2009 and March 2013, 128 sampling visits were completed. Following a break in sampling, a further 77 visits were completed between April 2014 and December 2017, giving rise to a dataset comprising 205 samples. A description of the sampling undertaken between 2009 and 2013 can be found in BEEMS Technical Reports TR120, TR196, TR215, and TR270, and details of the 2014 – 2017 sampling can be found in BEEMS Technical Report TR339. The dataset was used to provide an annual estimate of the numbers and weights of fish and invertebrates impinged at the SZC station for the 9-year period. A total of 91 finfish and 62 invertebrate taxa were recorded.

2.1 Selection of key taxa for SZC impingement assessment

The impingement assessment process for SZC is described in Section 5.

For the purposes of the Sizewell marine ecology impact assessments, taxa are key in the ecosystem if they meet at least one of the following criteria:

- Socio-economic value: Species that contribute to the first 95 % of the first sale value of commercially -landed finfish in the area off the east Anglian coast and contributes to the first 95 % of total abundance in at least one of the available datasets (2 m beam trawl, otter trawl, BEEMS eel survey, annual impingement). Commercial landings are recorded using statistical rectangles that divide the southern North Sea into areas of 30 minutes latitude by 1 degree longitude and covering approximately 900 nautical miles^2. For the purposes of describing local commercial fisheries, 6 rectangles have been considered, that extend from north Norfolk to the Thames estuary and eastwards to the middle of the North Sea (BEEMS Technical Report TR123). Socio-economic value was calculated using data supplied by the Marine Management Organisation (MMO) and which was used in BEEMS Technical Report TR123. 6 taxa (**Herring, bass, sole, cod, plaice, thornback ray**).

- Conservation importance: The "S41 Priority Species" spreadsheet given by Natural England (http://publications.naturalengland.org.uk/publication/4958719460769792) was used to assess the conservation status of the fishes recorded in the Greater Sizewell Bay. This spreadsheet was built based on the legislation in Section 41 of the Natural Environment and Rural Communities (NERC) Act 2006. It is worth noting that measures in place to provide protection for the named species apply to the adult stock rather than the eggs or larvae, and focus on halting the decline of the spawning stock biomass mainly via restriction on exploiting recruited species. The resulting list contains one species which has not been detected in the extensive BEEMS sampling programmes (Atlantic salmon). 16 taxa (**allis shad, twaite shad, European eel, herring, Atlantic cod, whiting, plaice, sole, salmon, sea trout, cucumber smelt, river lamprey, sea lamprey, tope, mackerel, horse mackerel).**

- Ecological importance: If a taxon is present in at least 30 % of samples and contributes to the first 95 % of total abundance in at least one of the available datasets (2 m beam trawl, otter trawl, eel surveys, annual impingement), we consider it to be common and/or abundant enough to play a key trophic role within the ecosystem. 13 taxa (**sprat, herring, whiting, bass, sand goby, sole, dab, anchovy, thin lipped grey mullet, flounder, cod, plaice, thornback ray)**.

There are 24 key fish taxa in the Greater Sizewell Bay in total based on either their commercial value, their ecological importance, or their conservation status. Several taxa fall under more than one criterion and four taxa are important with respect to all three (Dover sole, herring, cod and plaice) (BEEMS Technical Report TR345).

The 24 species are representative of the fish assemblage at Sizewell because:

a. they represent an average of 94.6% of the total fish impingement numbers during the CIMP programme from 2009-2013;

c. they contain examples from all functional guilds with the exception of freshwater species which, as would be expected, are rarely found at Sizewell;

d. they contain examples from all the feeding guilds and habitat groups;

e. they contain all of the indicator species found in the vicinity of Sizewell that are assessed in the WFD “fish” biological quality element in transitional waters; and

f. they contain the key prey species that supports the food web at Sizewell (including for HRA protected marine birds.

2.1.1 Conservation species impingement data

In the 9 years of extensive impingement sampling, catches of 4 conservation species were extremely rare or non-existent:

- Atlantic salmon 0 fish
- Allis shad 1 fish in 2009
- Sea lamprey 1 fish in 2015

- Sea trout 1 fish in 2010

None of these species were caught in any of the BEEMS fishing surveys. Salmon is, therefore, not expected to be impinged at SZC.

The lone Allis shad is considered to be a straggler from the Garonne population (BEEMS Scientific Position Paper SPP071/s) because this is the largest self-sustaining population that is closest to Sizewell. (There is no evidence that the Allis shad found in the Tamar are part of a self-sustaining population). Making no allowances for the fact that only one fish was caught, the predicted unmitigated impingement at SZC is five fish per annum, which decreases to only two fish per annum with the effect of the intake head design. It is not statistically valid to extrapolate this data point to future years and a more meaningful assessment would be to compare the scaled-up impingement in 2009 with the stock estimate for that year of 27,397 adults (BEEMS Technical Report TR456). This is the stock comparator that has been used in this report.

The predicted impingement for sea lamprey is also five fish but with the effect of the LVSE intake head design, this drops to only two fish impinged. Actual impingement losses considering survival through the fitted FRR mitigation are < 1 fish per annum (again using a statistically invalid extrapolation technique from the one fish caught in the CIMP programme). Allowing for the fact that only one fish was caught in the eight years when sampling took place, the predicted impingement is <0.13 fish per annum which is ecologically insignificant. No attempt has been made to put such a negligible impact into an adult stock context as the effect is insignificant.

The lone sea trout caught in the CIMP programme is considered to be from the UK North East coast population. A highly conservative assessment has been made in this report for sea trout based upon scaling up the 1 sea trout caught in May 2010 to a model output of 10 fish per annum for SZC without impingement mitigation. After adjustment for the impingement reduction by the LVSE intakes, the annual impingement loss estimate for SZC is approximately 4 fish per annum (Table 2). This estimate has then been compared with the annual mean net catch for sea trout in the sampling period to produce an estimated SZC effect of 0.01% of annual landings. The use of landings statistics overestimates the effect on the stock as landings are much less than the SSB. However, this simplistic assessment is not appropriate for such rare impingement events as it does not take into account that no other sea trout were detected in the period 2009 -2017. Allowing for the fact that only one fish was caught in the eight years when sampling took place, the predicted SZC impingement is approximately 0.5 fish per annum which is ecologically insignificant.

2.2 Aims of this report

The purpose of this report is to provide predictions of SZC impingement, based on the CIMP dataset collected between 2009 and 2017.

Aims:

- Provide predictions for all key finfish and shellfish species, based on the proposed design of the station with and without the selected impingement mitigation technology fitted.
- Place the predicted losses of these key species into the context of the most relevant stock unit area.
- Determine whether SZC impingement represents a significant effect for any of the key species
- Determine whether SZC entrapment (impingement predictions from this report +entrainment predictions from BEEMS Technical Report TR318) represents a significant effect for any of the key species
- The impingement predictions from this report are used to provide the source term for modelling the impacts of dead fish discharged from the SZC FRR system upon the local marine environment.

3 Selection of impingement mitigation measures for SZC

The Environment Agency have issued guidelines for the types of measures that could be adopted at new direct cooled power stations to reduce the predicted environmental impacts of impingement and indirectly the potential for pollution by discharges of dead fish back into the marine environment (Environment Agency 2005,2010). As explained in these guidelines, in practice the selection of potential impingement mitigation measures involves a complex consideration of the likely effectiveness of each measure in the marine environment at the station location, engineering feasibility and operational safety for staff and the plant. The range of feasible mitigation options is much larger in a freshwater and some brackish environments that do not present a high biofouling risk to station plant and for small abstractions of a few cumecs but many of these options are infeasible for a coastal direct cooled power stations (Environment Agency 2005). SZC's intakes would require an abstraction of 132 cumecs and would be mounted on the seabed in a highly turbid, coastal environment with high wave exposure offshore of the Sizewell-Dunwich Bank. The site is classified as being at high risk of biofouling.

3.1 Biofouling and other blockage risks

Design decisions on the choice of impingement mitigation options have to be taken in light of the environmental risks to the plant from the Sizewell marine environment. The cooling water system for SZC would be nuclear safety classified and it is therefore extremely important that the system is designed to prevent blockages of critical plant. The blockage hazards at Sizewell include:

- marine debris (discarded nets and ropes and marine litter – this risk also includes potential impact damage from large and heavy items) and large clogging organisms (e.g. in the southern North Sea: sprat shoals and ctenophore blooms)
- colonisation by biota (biofouling) that could cause blockages and subsequent reductions in cooling water flow in the system (e.g. shellfish, barnacles, reef forming organisms)
- siltation due to high suspended sediments (risks designed out by elimination of low velocity regions in high risk zones of the cooling water system e.g. intake heads and forebays

Established power station design practice is to progressively reduce the risks from marine debris and large clogging organisms by the use of robust, coarsely spaced intake bars at the intakes followed by two tiers of filtration within the plant (trash racks and then drum or band screens). Colonising organisms are deterred by chlorination of the cooling water system (Note chlorination is designed to deter settlement rather than to kill organisms).

EDF Energy's policy for its existing UK fleet is that stations exposed to a high biofouling risk should have the capability of maintaining a default regime of continuous, year-round chlorination to obtain 0.2 mg l^{-1} Total Residual Oxidant (TRO) in the discharge water from plant vulnerable to biofouling. Sizewell B is currently assessed as subject to a high risk of biofouling and operational practice is, therefore, to maintain the default regime. The detailed application of the EDF Energy policy, for example whether the entire cooling water (CW) system is dosed continuously or just critical plant, is dependent upon site specific issues such as the flexibility of the chlorination plant. At SZB the current policy is to dose the entire CW system, including the inlet tunnels, throughout the year.

Based upon the known risk of biofouling at Sizewell, it would be necessary to dose critical plant at Sizewell C (the condensers and essential cooling water systems) during the growing season when seawater temperatures exceed 10 °C and also to have the flexibility to dose those systems at other times of the year based upon operational need. The chlorination policy for the other parts of the CW system has to be effective against any biofouling risk that would threaten the operation of the station whilst minimising toxicological effects on non-target species. In particular, Sizewell C will be fitted with a Fish Recovery and

Return (FRR) system (Section 3.4.1) to reduce the mortality of impinged fish. The Environment Agency best practice screening guidelines are that, wherever possible, chlorination should be avoided before the FRR so as to minimise any loss of fitness for those fish returned to the marine environment. This is a larger environmental issue for Sizewell C (SZC) than Sizewell B (SZB) due to the length of the SZC CW tunnels and the potential significant increase in TRO exposure time, dependent upon chlorination system design, for organisms abstracted into the CW system.

3.2 Impingement Mitigation Optioneering

EDF Energy have carefully considered each of the available options in the Environment Agency guidelines. The initial stage is to determine the location of the intake and outfall locations at a site based upon environmental and engineering considerations. In practice given practical engineering constraints on intake siting and intake tunnel length, the available options may not significantly change the predicted abstraction of organisms in a well-mixed environment. For most species this is the case at Sizewell (Section 5.1.1). There are measures to reduce the mortality of impinged fish and crustacea but for planktonic organisms (e.g. zooplankton, fish eggs and fish larvae) due to their small size (typically less than a few mm in length or diameter) little can be done to reduce the numbers and mortality of entrained organisms. However, due their number, spatially ubiquitous nature and the high natural mortality of the majority of planktonic organisms, entrainment impacts from coastal power stations are rarely significant and that is the case predicted for SZC (Section 7.7). There are two complementary mitigation technological approaches to minimise impingement losses using:

a. Biota exclusion technology - measures to minimise the number of organisms abstracted into the station intakes; and

b. Biota recovery technology - measures taken inside the cooling water system to filter organisms out of the cooling water stream and safely return as many as possible of them alive to sea.

3.3 Biota exclusion technology

Several techniques are available, with variable effectiveness, to reduce the number of fish and crustacea being abstracted with the seawater and impinged on the cooling water fine filtration systems (drum or band screens). These measures are located at or close to the intakes which for SZC would be in the open sea at more than 3km offshore to the east of the Sizewell-Dunwich sandbank and mounted on the seabed. From a design perspective the five key requirements of such technology for SZC are:

- Compatible with nuclear safety requirements for an uninterrupted supply of cooling water for the 60-year operational life of the station. This implies the use of systems that are highly resistant to damage or blockage and that are readily maintainable in all weather conditions, all year round.
- Operation and maintenance compatible with the EDF Energy’s zero harm safety policy for staff and contractors. i.e. no requirement for activities judged hazardous to human life.
- Proven operational experience in a similar environment that demonstrates reliable delivery of effective environmental mitigation.
- Due to the offshore environment any system should preferably use entirely passive technology e.g. requiring no power, chemical supplies or compressed air systems that could compromise reliability and hence nuclear safety and environmental effectiveness.
- System operational maintenance requirements must be compatible with high power plant availability.

The potential biota exclusion techniques include:

i. Physical barriers (e.g. wedge-wire screens and bubble curtains);

ii. Auditory or visual behavioural deterrents that aim to deter fish from a trajectory likely to cause their abstraction (e.g. acoustic fish deterrents (AFDs) and strobe lights; respectively).

iii. Design of the intake heads to minimise the risk of biota abstraction e.g. by use of velocity-capped intake heads, limiting intake velocities, minimising abstraction cross sectional area by mounting the intake orthogonal to the tidal flow. Low velocity side entry (LVSE) intake heads combine all of these attributes and represent the state of the art for such an approach and are included in the approved design for HPC.

Several of these measures have been deployed in riverine locations. However, for existing coastal power stations only limited intake head design improvements (e.g. use of velocity caps, and sizing intakes to reduce intake velocities) or more recently AFDs have been deployed. AFDs have only been deployed at stations with onshore or very nearshore intake locations and none have been deployed at far offshore locations (e.g. the 3km+ offshore locations proposed for SZC).

Each of the biota exclusion options are considered in turn below.

3.3.1 Passive wedge-wire cylinder screens (PWWC screens)

In principle, the best form of impingement mitigation could be to prevent abstraction of fish and crustacea by the use of very fine screens with gaps of a few millimetres. Environment Agency (2005) considers that "Passive wedge-wire cylinder (PWWC) screens are a tried and tested solution and are generally regarded in Britain as the best available technology for juvenile and larval fish protection". Wedge-wire' refers to the cross section of the welded wires that are wound helically to form a cylindrical screen surface.

PWWC screens are commonly used to exclude fish from small, riverine abstraction intakes and there are also some small industrial applications in brackish and saline waters. However, because of the very small gaps between the wires the screens are at high risk of becoming blocked or damaged by floating debris, weed and litter even in riverine environments. In the marine environment the risks are much greater due to the potential hazards from ctenophore blooms, pelagic fish shoals and marine debris and also particularly from biofouling by colonising organisms. To reduce the blockage risk in high risk environments, PWWC systems have complex maintenance requirements with frequent, active cleaning required (for example with rotating mechanical brushes or by the injection or high-pressure air). The maintenance advantage of the passive screen then largely disappears as the active cleaning systems present significant reliability risks. Practical PWWC systems are complex systems, that must be recovered for regular maintenance including to repair damage to the fine screens. To do this such filters are usually track mounted on large motorised platforms attached to the shore.

It is instructive to consider the theoretical sizing of a PWWC system for SZC. The abstraction capacity of PWWC filters depends upon the wire spacing. To eliminate most impingement issues (and to comply with the Environment Agency screening guidelines for glass eels) it would be necessary to employ a 2 mm wedge wire filter. The largest commercially available 2 mm PWWC filters would permit an abstraction of 2.7 cumecs with an 8m long, 2.5m diameter cylindrical filter i.e. to create the required 4 intake head system for SZC with a total abstraction of 132 cumecs would need a minimum of 13 PWWC filters per head, probably15+ allowing for redundancy for cleaning and repair. i.e. a total of up to 60+ filters for SZC with each head length being greater than 130m in length, all track mounted on 4 powered offshore platforms complete with compressed air cleaning and antibiofouling chemical supply. The lifetime for such complex systems are unknown but certainly are not the required 60 years for SZC. The required recovery frequency for the filters is also unknown but in the growing season could be monthly. The filters would require frequent replacement using heavy lifting equipment and the track mounting would also require replacement, probably on at least a decadal frequency. No such system has been deployed at any power station worldwide.

Conclusion: PWWC filtration is an unproven technology for direct cooled nuclear power stations and is not considered compatible with nuclear safety requirements for a constant supply of cooling water.

This is the only potential technology that could theoretically eliminate impingement and therefore any other mitigation measure would require technology to recover organisms from the station drum and band screen.

3.3.2 Bubble curtains

The use of a bubble curtain, whereby air is released along a section of seabed to create a ‘curtain’ of bubbles rising to the surface and thus creating a barrier to fish, is a potentially useful exclusion technique in still or slow-moving waters but is not suitable as a permanent exclusion system for waters where tidal currents ebb and flow at >1m/s and would break down the curtain of bubbles. Small sized bubble barrier systems to reduce underwater noise have been temporarily deployed at sea from jack up rigs during periods of low tidal velocities during windfarm piling activities but never around the required large intake heads of a power station at all states of the tide and in all weather conditions.

Conclusion: bubble curtains are an unproven technology for use around the offshore intakes of direct cooled nuclear power stations and are considered unlikely to deliver substantial impingement mitigation at Sizewell.

3.3.3 Behavioural deterrents – strobe lights

Strobe lighting, can be used to deter some fish (for example eels), however, their effectiveness in turbid coastal waters is unproven (turbidity relates to the amount of material; suspected in the water and thus restricting visibility). The surface water at Sizewell is classified as “intermediate turbidity” and experiences increased levels of turbidity in autumn to spring when storms and increased wave action stirs up sediment. Repeated surveys have demonstrated that it is extremely difficult to photograph any seabed features at Sizewell due to this limited visibility. The near-bed conditions at the Sizewell C intake locations have particularly high levels of suspended sediment with measured levels of greater than 2 g/l at the height of the proposed intake surfaces due to sediment transport around the Sizewell-Dunwich Bank and mean winter values of ca. 500 mg/l (BEEMS Technical Report TR498). Such high levels of suspended sediments dramatically reduce the penetration of light through the water and would prevent strobe lighting from having an effective deterrent function.

Conclusion: strobe lighting is an unproven technology for use around the offshore intakes of direct cooled nuclear power stations and is considered unlikely to deliver impingement reductions at Sizewell.

3.3.4 Behavioural deterrents - electric barriers

An electric fish barrier is a non-physical barrier that prevents fish passage from one location to another or induces fish movement from one area to another within a body of water using an electric current. Electric barriers pass an electrical current through the water, thus creating an electric field. As fish enter the electric field they become part of the electrical circuit and experience electric current flowing through their body. As the fish approaches the anode, the electric field intensifies, which causes the fish to generally turn around and swim away from the electric barrier. The set-up of an electric barrier requires a series of electrodes, alternating anodes and cathodes to span across a body of water. However, electric barriers are affected by water conductivity and are unsuitable for marine or brackish water environments, therefore, the use of an electric fish barrier is not feasible for the offshore Sizewell C intakes.

Conclusion: Not feasible

3.3.5 Behavioural deterrents - Acoustic fish deterrents (AFDs)

Well-designed AFD systems have been reported to reduce impingement of some fish species by creating high intensity sound fields of swept frequency pulses of sound around an intake thereby causing some fish to change direction and move away from the sound field. The available commercial systems are predicted to work well with sensitive pelagic species (e.g. sprat and herring), moderately for a range of demersal species (e.g. cod, whiting) and poorly for species with either low hearing ability or low responsiveness to the underwater frequencies used such as eels, lampreys, and some flatfish. All power station installations of AFDs are either on shore or very close to shore to facilitate system maintenance. There are no AFD systems operating in environments at multiple kilometres offshore as they would have to be at SZC.

An AFD system for SZC would necessarily be very similar to that evaluated for HPC and would require up to 288 underwater sound projectors located at the CW intakes approximately 3km offshore. The issue of system longevity is a particular constraint as the AFD sound projectors need to be recovered and serviced on shore currently at 12 monthly intervals, possibly at up to 18-month intervals with further research. Other key issues with installation of AFDs in offshore environments are the large number of electrical components required at each intake, the supply of reliable high levels of electrical power and control telemetry to the individual sound projectors and the required close proximity of the projectors to the intake heads themselves (without affecting the hydrodynamic performance or structural integrity of the head. The recovery of the projectors for maintenance would be a major issue. Four permanent offshore platforms with track mounted projectors could theoretically be used but the these would have to be very close to the intakes, affecting the hydrodynamic performance of the intake heads and the resultant sound field would be likely to have an unpredictable deterrent effect due to complex interference patterns caused by the structures themselves. More significantly, such a complex track mounted system with a profusion of electrical wiring would also have major issues with long term reliability for the 60-year lifetime of the station in the corrosive marine environment at the site. The use of such structures is not considered feasible. In the near zero underwater visibility at the SZC intakes the use of robotic servicing via remotely operated vehicles is not considered feasible and instead each of the projectors would have to be recovered and replaced using divers working by feel, operating from anchored support vessels for months during every year, for the 60-year lifetime of the station. The system would be extremely complex to construct and to maintain with offshore operations restricted to narrow tidal windows and subject to lengthy periods of weather downtime in the exposed location offshore of the Sizewell-Dunwich Bank. An assessment of the risks involved with such an operational system has concluded that the safety risks to maintenance staff would be unacceptable.

Conclusion: Logistical and safety considerations preclude the use of AFDs at Sizewell C.

3.3.6 Intake design - LVSE intakes

Low velocity side entry (LVSE) intake heads have already been designed and received regulatory approval for use at HPC. These very large intake structures are designed to minimise impingement by:

a. reducing vertical velocities which fish are ill equipped to resist by means of velocity caps on the intakes.

b. limiting the exposure of the intake surfaces to the tidal stream and in so doing reduce the risk of impingement for fish swimming with the tidal stream. i.e. to reduce the cross-sectional area of the intake to the prevailing tidal directions by mounting the head orthogonally to the tidal flow.

c. reducing intake velocities into the head to a target velocity of 0.3m/s over as much of the length of the intake surface which will maximise the possibility of most fish avoiding abstraction

LVSE intakes have the advantage of reducing impingement for all fish species at risk of abstraction. The HPC LVSE intakes will be the first deployment of this technology on an operational power station worldwide and represent a considerable advance in the design of intake heads. One option for SZC would have been to reuse the HPC design. However, Hinkley Point has a low biofouling risk whereas Sizewell has a high risk and the starting point for SZC would be an assumption of chlorinating the entire cooling water system starting at the intake heads. Studies demonstrated that the required chlorine dose combined with the exposure time in the 3km tunnels would significantly reduce the survival of many of the species that the FRR is designed to protect. After a consideration of risk and engineering feasibility by EDF Energy, based upon operational experience at Sizewell, the size of the proposed SZC intake tunnels and the use of a simple SZB style capped intake for SZC it was decided that it would be acceptable not to chlorinate the intakes and intake tunnels, thereby significantly reducing the exposure of abstracted organisms to TROs. This led to the initial recommended chlorination policy for SZC described in BEEMS Technical Report TR316:

a. Maintaining a TRO level of 0.2mg l^{-1} at the discharge of critical land-based plant (condensers and essential cooling water systems) throughout the year.

b. Intake heads, inlet tunnels and the forebays not to be chlorinated because of the impracticality of operational control of the chlorination dose in the intakes and inlet tunnels and so as not to compromise the FRR system effectiveness.

c. Velocity-capped intake heads of a similar design to Sizewell B to be employed at Sizewell C. Such intake heads are much more readily maintained and much less likely to biofoul than the low velocity side entry (LVSE) intake heads planned for Hinkley Point C.

d. In order to protect the drum screens and FRR system, chlorinate the drum screen wells but only in the growing season when seawater temperatures exceed 10 °C.

e. Apply for a WDA discharge permit for TRO (measured before the discharge tunnel) of 0.2mg l^{-1} throughout the year.

Further studies demonstrated that even this revised chlorination policy would impair the effectiveness of the FRR system for demersal species such as the juvenile bass found at Sizewell. After a further careful engineering review the first dosing point in the cooling water system was moved to after the drum screens thereby removing chlorination from the FRR system.

The decision to use SZB style omnidirectional velocity-capped intakes was based upon a risk assessment that concluded that the unchlorinated LVSE intake heads designed for HPC would present an unacceptable biofouling risk at Sizewell due to the surface area of the baffles inside the intake head structure. If the HPC LVSE heads were fitted at SZC the entire CW system would have to be chlorinated, effectively eliminating the benefit of the FRR system for most species. However, the SZB style velocity-capped intake heads would offer no impingement rate improvement over the existing SZB intakes; in particular organisms swimming in the tidal stream would still experience high intake velocities in excess of 1m/s for a large part of the tidal cycle which would be too high to permit most organisms to avoid abstraction.

To reduce the biofouling risk, it was necessary to remove as many of the internal baffles in the HPC LVSE intake design as possible and to reduce areas of low velocity flow within the head. After extensive computational fluid dynamics (CFD) modelling studies of a range of LVSE designs it was determined that a modified version of the HPC LVSE intake heads could be designed to have low biofouling risk and achieve the same reduction in cross sectional area as the HPC intake heads. By improving flow dynamics within and around the heads the modified heads the variation in intake velocities across the intake surfaces could also be minimised and permit low intake velocities to be achieved over the whole tidal cycle (Section 7.8.3, BEEMS Scientific Position Paper SPP099).

Conclusion: LVSE intakes, modified from the HPC design to reduce biofouling risk would provide reliable and effective impingement reduction for SZC.

3.4 Biota Recovery Technology

3.4.1 Fish Recovery and Return System

A Fish Recovery and Return (FRR) system is designed to return robust species (particularly flatfish, eels, lampreys and crustacea and to a lesser extent demersal species such as bass, cod and whiting) that are impinged onto the station drum and band screens safely back to sea. A state of the art fish recovery return system has been designed and received regulatory approval for Hinkley Point C. This system has been subject to intensive design scrutiny and complies with Environment Agency guidelines for such systems. EDF Energy policy for SZC is to replicate the design of HPC as far as possible and so it has been decided to incorporate the HPC design into SZC. The tidal range at Sizewell is less than at Hinkley Point and it has, therefore, been possible to improve the fish friendliness of the SZC FRR system by removing an Archimedes screw system that is essential to manage water levels in the HPC cooling water system.

The drum and band screen employ fine mesh filters to remove impinged organisms from the cooling water flow. The default mesh size for the EPR reactor is 5 mm square as opposed to the 10 mm mesh filters employed at SZB. After careful consideration of the risk of clogging by summer ctenophore blooms at Sizewell, it has been proposed to fit 10 mm mesh filtration for SZC which has been operationally proven not to cause clogging at SZB. Trash racks are used to protect the drum and band screens from damage and over loading. With a 5 mm mesh the track rack has to have 50 mm vertical bar spacing, a 10 mm mesh size could allow the trash rack bar spacing to be increased to 75 mm. The environmental effects of the larger track rack bar spacing and the larger mesh size are beneficial and will reduce the predicted effects on fish stocks. The larger mesh size will also reduce discharges of dead biomass from the FRR outfall. There would be no adverse effect on the survival of any glass eel that may be abstracted as glass eels would pass through both 5mm or 10mm filtration.

Conclusion: FRR systems with proposed 10 mm mesh screens would prove reliable and effective impingement reduction for SZC.

3.5 Conclusions

Studies of impingement reduction technologies for SZC demonstrated that two measures were both feasible and likely to reliably deliver reductions in the predicted losses of fish and crustacea. These are LVSE intakes and an FRR and both of these technologies are planned for the proposed SZC station.

With these measures fitted at SZC, the predicted reduction in impingement mortality compared with an unmitigated station based upon the expected performance of the LVSE intakes (Section 5.6) and the FRR system (Table 6) are shown in Table 3.

Table 3. Predicted reduction in impingement mortality for SZC fitted with LVSE intakes and FRR system compared.

Group	Example species	Impingement reduction
Pelagic fish	sprat, herring, anchovy, shad	62%
Demersal fish	bass, cod, whiting, grey mullet	77-79%
Epibenthic fish	eel, lampreys, sole, sand goby	92%
Shellfish	Crab, lobster, brown shrimp	92%

4 Relevant site features

The Sizewell site hosts SZA and SZB. SZA ceased operation at the end of 2006 and is being decommissioned. SZB is a direct-cooled nuclear power station using a pressurised water reactor design to generate an electrical output of about 1195 MW. The SZB intake and outfall structures are located inshore of the Sizewell Bank system (Figure 1). For SZB the volume of water extracted is 51.5 $m^3 s^{-1}$ (51.5 cumecs). The intake structure consists of two intake heads, each initially leading to its own tunnel, which then join to form a single tunnel. Each head is octagonal and ~11.5 m across and is omnidirectional. The structure sits ~1.5 m above the seabed and the intake aperture is 3 m high. The tidal flows in the region are highly rectilinear and peak at 1 m s^{-1} on spring tides. During peak tides, a tidal streamline the width of the intake enters the intake, whereas at slack water the water is drawn from a radius around the intake.

The SZB intake was designed to not include any large superstructure that might attract fish and has a cap to limit drawdown from the surface. There are no screen bars or other devices that are designed to reduce fish entrainment or impingement at the intakes. There is also no provision for maintenance or internal access. The intake tunnel consists of nine square precast concrete caissons, each 4.82 m wide (internal) and 702 m long. Flow velocities in the tunnel are approximately 2.5 m s^{-1}, giving a passage time through the tunnel of approximately 5 min. The capped intake design was intended to ensure that warm, surface water was not drawn into the station, as well as to reduce the likelihood of there being a surface vortex. However, the capped design, by eliminating vertical water movement, was also expected to help fish avoid entrapment into the intakes, because fish are ill-equipped to respond to vertical water movements. Velocity caps were expected to be especially protective of pelagic species such as sprat and herring. Studies undertaken in March/April 1994 concluded that the B station impinged significantly fewer fish than the A station, which was not fitted with a velocity cap (Turnpenny and Taylor, 2000). The SZB discharge is through two tunnels that point offshore and upward with the discharge in the nearshore region ~150 m from the low tide mark (Figure 1).

The proposed SZC intakes will be low velocity side entry (LVSE) structures designed to reduce fish impingement. A total of 4 LVSE heads will be fitted with two heads fitted on each of the two intake tunnels. The intake heads will located approximately 3km offshore, on the eastern side of the Sizewell-Dunwich Bank in approximately 15m (ODN) depth (Figure 1). The detailed design of the intake heads is not yet complete but the performance of the design with the worst hydrodynamic performance has been used for this impingement assessment in order to envelope worst case impingement effects. This design has intake surfaces of the same size as those planned for Hinkley Point C (HPC) with the bottom of the intake surfaces being at more than1m off the seabed (as at HPC) in order to reduce abstraction of benthic organisms. The preferred outfall locations 09a and 09b marked are contingent on final engineering geotechnical assessments but will be there or slightly further offshore and in approximately 18m depth.

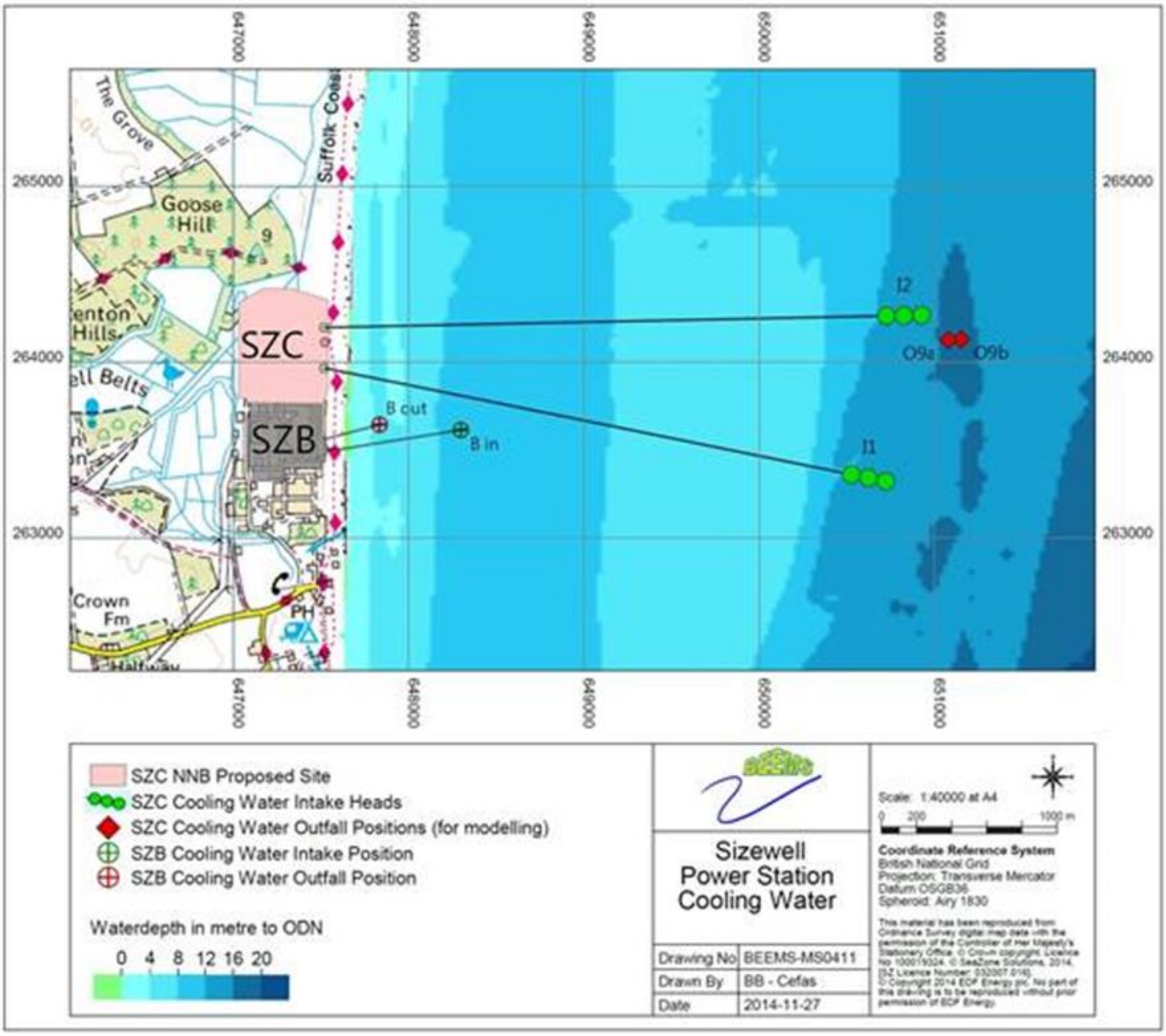


Figure 1 The coast at Sizewell, showing the locations of the intake and outfall for SZB and the proposed intakes and outfalls for SZC. The locations of three intake locations are shown for each SZC tunnel but only 2 heads will be fitted per tunnel with locations dependent upon geotechnical considerations.

5 Impingement Assessment Methodology

5.1 Introduction

To estimate the unmitigated impingement at SZC the assessment approach adopted in this report is to scale the measured impingement at SZB by the ratio of the cooling water volumes extracted by the two stations. The accuracy of the assessment depends upon whether:

- the fish community is the same at the location of the SZC intakes (approximately 3 km offshore) as at the SZB intakes (approximately 700 m offshore); and
- the SZC intakes will abstract the same amount of fish per cumec as SZB.

5.1.1 Differences in the abundance of fish at the SZB and SZC intake locations

The results of subtidal fishing surveys in the greater Sizewell Bay area are described in BEEMS Technical Report TR345). Ten demersal fishing surveys were carried out over a 4-year period. Sampling was conducted using two different fishing gears – a 2 m beam trawl and a commercial otter trawl. A coastal pelagic fish survey was also carried out in March and June 2015. These surveys found predominantly juvenile fish. Forty species were identified in the 2 m beam trawl catches, 25 in the commercial otter trawl catches. These fishing surveys found no significant spatial differences in the fish community nor the fish length distributions between the locations of the SZC and SZB intakes.

5.1.1.1 Bass distribution within Sizewell Bay

Bass impingement at SZB consists mostly of juvenile fish with few mature adults. From the CIMP survey from 2009- 2017 it is known that bass impingement at SZB takes place almost exclusively in winter (with 98.8% in the period November to March and 1.2% from April to October). In the period of lowest seawater temperatures (December to March) bass impingement is 96.1% of the annual total). It is well known that juvenile bass are attracted to the warm water outfalls of power stations in winter (Jennings *et al* 1991) and this has led to the creation of areas of restricted fishing around several UK power station outfalls. It was, therefore, expected that there would be differences in bass spatial density at Sizewell due to the SZB discharge but the extent of the area of attraction at Sizewell was unknown. To resolve this issue a highly targeted survey programme was undertaken to investigate bass distribution in the Greater Sizewell Bay (BEEMS Technical Report TR380). The aim of the programme was to quantify the impact of the SZB discharge upon the spatial distribution of bass and specifically to determine how the abundance of bass differed inside and outside of the SZB plume and whether there was a significant difference in abundance at the locations of the SZB and proposed SZC intakes.

In February 2016 a 5-day sampling programme was undertaken using an otter trawl known to be efficient at sampling bass. The high trawl opening allowed the sampling of fish from a larger part of the water column than a beam trawl (from just off the bottom to approximately 1m off the seabed) and the 4 mm mesh liner retained small individuals. February was chosen as this corresponds to the period of peak bass impingement at SZB (43% of the annual impingement total). Sampling was undertaken inside and outside of the Sizewell-Dunwich Bank, and close to and distant from the current and proposed intake/outfall locations of SZB and SZC, respectively. Forty-one tows were completed in the Greater Sizewell Bay. Three sampling sites were located outside the Sizewell-Dunwich bank (“offshore”) in similar depth strata and habitats. One site was in the area of the proposed SZC intake/outfall structures and the other two were some distance north and south of this. Five sites were located inside the Bank (“inshore”); the first immediately north of the SZB outfall; two about 1 km north and south of this, but within the SZB thermal plume; and two several km north and south and outside the plume. During the survey, 110 bass were recorded, ranging between 15.5 and 45 cm TL, with the majority being 2 y old (70.9%) followed by 3 y olds (14.5%) and the remainder up to 6 years of age (No 0 or 1 group fish were caught). The survey found:

a. A statistically significant majority of bass at Sizewell (105 bass or 95% of the bass caught) were recorded inshore of the Sizewell-Dunwich Bank.

b. A statistically significant increase in bass abundance was found near the SZB plume inshore of the Bank compared with other inshore stations to the north and south of the SZB outfall that were effectively outside of the influence of the SZB plume.

c. A statistically significantly greater abundance of herring and sprat were observed inshore of the Bank than offshore. There was no significant difference in the distribution of whiting across the Bank.

The sampling programme therefore confirmed previous observations that bass are attracted to warm water in winter. In particular, the abundance of bass at Sizewell was significantly different inshore and offshore of the Bank which would be expected based upon the significant difference in water temperatures near to the seabed at the two locations. The inshore sampling also demonstrated that the abundance of bass was significantly greater inside of the SZB plume than outside. These results provide confidence that the observed bass distribution was strongly associated with the presence of the SZB discharge plume and this has implications for the expected impingement rates for SZC which are discussed in Section 7.5.

5.1.2 Effect of the proposed SZC intake heads upon the expected impingement rate

SZC would be fitted with 4 low-velocity side-entry (LVSE) intake heads, mounted on the seabed approximately 3 km offshore. The proposed intake heads would be capped structures with the intake surfaces orthogonal to the direction of the tidal flows. The intakes are specifically designed to reduce the cross-sectional area available to intercept any fish being transported in the tidal flows. The reduction in cross-sectional area combined with the low intake velocity is predicted to substantially reduce the number of fish abstracted per cumec of abstracted seawater compared with SZB which is fitted with conventional omnidirectional intake heads with intake velocities that exceed the ability of most fish to avoid being abstracted. Modelling indicates that the SZC station will abstract 0.383 per cumec of the fish abstracted by SZB, because of the intake head design. The methodology used to derive this impingement reduction factor is described in BEEMS Scientific Position Paper SPP099 and is based upon the methodology proposed by the Environment Agency for impingement studies at Hinkley Point C.

The effect of the intake head design on the abstraction of organisms, is assessed by first multiplying all predicted values for SZC (unmitigated), by the factor of **0.383**. The full CIMP impingement assessment process is illustrated in Figure 2

5.2 SZB impingement data collection and collation

Impingement sampling at SZB was initiated in 2009 to provide information that could be used to predict the losses of the proposed SZC station. The sampling scheme consisted of sampling for 6 * 1-h samples in the daylight in addition to one * 18-h sample that was collected overnight. In each sample, the impinged material was sorted to species where possible, weighed and the fish fauna were measured. If subsampling was required, the data were raised to the individual sample first, before all 7 samples (six hourly, one overnight) were summed to give an estimate of the 24 hours of sampling. A total of 128 sampling visits was completed between February 2009 and March 2013. Each sample represented the estimated number and weight of fish that would have been impinged during the 24-h period, if the station was working at full capacity (i.e. 4 pumps in operation, which is not always the case during the year). The impingement data for 2009-2013 are described in four separate annual reports (BEEMS Technical Report TR120; TR196; TR215; TR270).

Sampling resumed in April 2014 and is ongoing. Prior to April 2014, the sampling methods were reviewed, particularly the need for the overnight sampling, to ensure that sampling was still appropriate to the data requirements. In general, the same sampling methods were adopted when data collection re-commenced. The only differences to sampling methodology were a change to the order in which the hourly and overnight samples were collected, the use of an Electronic Data Capture system (EDC) to improve the efficiency and reliability of data capture, and a change from measuring fish using Standard Length, to using Total length. Sampling has continued since April 2014. Between 2014 and 2017, a further 77 samples were obtained, and these were similarly raised to represent 24 hours with the station pumping at full capacity. Details of the sampling review, revised sampling methodology and data handling are given in BEEMS Technical Report TR339.

Both data sets were brought together to provide a single data set of 205 sampling visits, each providing information on the number and weight (and number at length for fish), of individuals impinged by the SZB station in 24-h at full capacity (BEEMS Technical Report TR339).

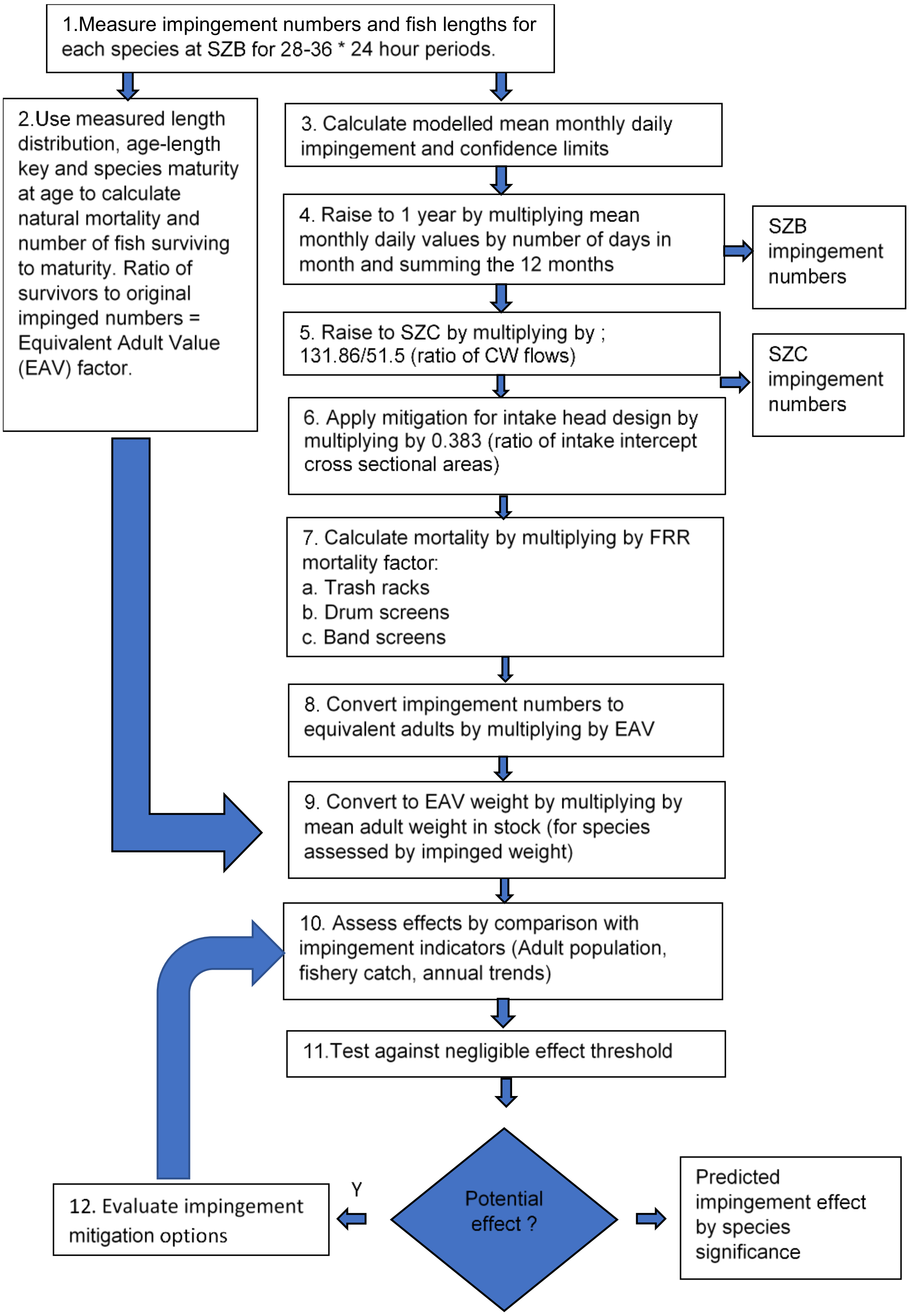


Figure 2 The SZC CIMP impingement assessment process

5.3 Estimated annual mean, minimum and maximum losses by the SZB station

Estimates of impingement for Hinkley Point C were based on a CIMP conducted at HPB in 2009 – 2010. For that dataset, sampling was conducted over 40 visits throughout the year. Estimates of annual impingement were calculated by first summing all samples from each quarter of the year and raising the quarterly total by the ratio between the number of days in the quarter and the number of sampling visits. The quarterly totals were summed to give an annual impingement estimate. Variability was estimated by bootstrapping the dataset – randomly selecting a subset of visits from each quarter and repeatedly estimating the total annual impingement to give 95 % confidence estimates around the mean value. This was possible because the number of samples available was evenly distributed thought the sampling year.

The SZB CIMP dataset is extensive, spanning 9 years and 205 samples. Ideally, with this time series, we would calculate and present estimates for impingement for each year separately, so that year-year variation could be observed. Impingement estimates for each year would be calculated using the same methods described for Hinkley Point, i.e. quarterly raising and the use of bootstrapping to provide confidence estimates. However, Hinkley Point B has dual reactors which means that even during outages there are rarely periods when the station is not abstracting seawater and this allows a near continuous impingement record. Sizewell B has a single reactor and therefore during outages no seawater is abstracted and therefore the impingement record is discontinuous and the Hinkley Point assessment methodology cannot be used. In particular, very little sampling occurred in 2013 (sampling was suspended in March 2013), and there are large sampling gaps particularly in the later sampling years (e.g. quarter 1 in 2014 and 2015). This was largely due to outages when no sampling could take place (Table 6). Therefore, raising the data quarterly would bias those species that show strong seasonal patterns in their impingement abundance (a frequent occurrence), leading to under or over-estimates of impingement. For example, due to the lack of samples in quarter 1 in 2014 and 2015, the numbers of bass and thin-lipped grey mullet recorded was small. Raising these years to an annual estimate would lead to significant underestimations of these two species in those years, which could be incorrectly interpreted as a decrease in abundance for those years.

Although the quarterly raising and bootstrapping approach could be used for some years with more evenly distributed sampling, it could not be consistently applied across the whole dataset. Further, there is no way of selecting any single year to represent impingement sampling at SZB. Consequently, a different approach was required to estimate annual impingement at Sizewell.

Table 4 Summary of the number of sampling visits to SZB completed between 2009 and 2017

	2009	2010	2011	2012	2013	2014	2015	2016	2017	Total
January		2	5	1	2			2	3	15
February	3	4	2	3	3			3	2	20
March	4	2	2	1	2			2	2	15
Q1 total	**7**	**8**	**9**	**5**	**7**	**0**	**0**	**7**	**7**	**50**
April	3	3	2	2		3	1		2	16
May	3	3	2	3		2	3		1	17
June	3		2	2		2	2	2	3	16
Q2 total	**9**	**6**	**6**	**7**	**0**	**7**	**6**	**2**	**6**	**49**
July	4	2	2	2		3	3	2	3	21
August	4	2	3	3		2	2	2	3	21
September	1	3		2		1	2	3	1	13
Q3 total	**9**	**7**	**5**	**7**	**0**	**6**	**7**	**7**	**7**	**55**
October		4	3	3			3	2	1	16
November	7	3	3	2			2	3		20
December	4	3	2	2			2	2		15
Q4 total	**11**	**10**	**8**	**7**	**0**	**0**	**7**	**7**	**1**	**51**
Total	**36**	**31**	**28**	**26**	**7**	**13**	**20**	**23**	**21**	**205**

5.4 Statistical method used to derive annual impingement estimates at Sizewell

Impingement estimates were made by fitting a statistical model (developed in R software) to the impingement data. Each species was assessed separately and there is no aggregated result by habitat or trait. Sampling was carried out on 205 different days between 4th Feb 2009 and 5th October 2017, and if a species was not encountered on a sampling day it was marked as an absence (0) for that day. For some species this resulted in a timeseries of mainly zeros.

The model chosen to characterise each species was different based on the number of occasions where an organism of that species was present. For species where presence was relatively continuous, a more complex model was applied which could assess how the number of organisms impinged varied from month-to-month and year-to-year (ZINB model). For species where absences were generally more common than presences, the model was restricted to investigating only the month-to-month variability as there was not enough data to investigate interannual variability (NB model).

ZINB models are known as Zero-Inflated Negative Binomial models and are computationally more complex because they consist of two parts to handle data with many zeros. Zero-Inflated Negative Binomial models generate a presence-absence model as well as an abundance model and combine the two models into one to return an estimate of abundance. This model asks how likely any presence is at a certain point in time, and how abundant is it likely to be at this point in time if it is likely to be present. Therefore, ZINB model models are the preferable tool for characterising a dataset with large number of zeros and the resulting estimate should be more accurate for this model than for NB models.

NB models are Negative Binomial models and are simpler because (a) they consist only of the abundance part of the model and (b) because they are not set up to look at the changes in abundance from year-to-year (interannual variability). This model therefore asks how abundant a species is likely to be during a certain month. The resulting estimate should be less accurate for this model.

Where possible, the ZINB model was used, but this was not always possible. For species where there were less than 22 out of the 205 sampling occasions, there were not enough presences for this model to be used since this model aims to identify both variability between years and variability caused by time of year. To do this, a relatively large (~10% or greater) number of presences is needed, and the presences need to be distributed amongst the various years which was not always the case. Where this is not the case, a ZINB model will not converge, meaning the effect of year and month cannot be obtained from the model.

If the ZINB model assessing the effect of year and month could not be used, the temporal resolution was reduced, making the model more likely to converge. The time data were passed to the model in coarser time periods. For example, instead of assessing each month as a separate time-step, each two months (one sixth of a year) was used. If there was still too little data for the model to converge, three months (one quarter of a year) was used as a time-step. If the model still did not converge (as was the case for any species where there were less than 22 out of the 205 sampling occasions), the simpler NB model was used. Therefore, within the ZINB models, there are 3 different levels of resolution depending on how much data was available: every month, every bimester, and every trimester. Each species was assessed to the best resolution possible, ideally monthly. All NB models used have a monthly resolution but did not assess the influence of year (no random variance caused by interannual variability).

Regardless of the timestep used, the influence of year was treated similarly for all ZINB models. A year was defined from February to the following January and was separated into two periods which were treated as separate events from February-July and August-January, contributing a random variance to the model, and to recognise that anomalously high abundances in one half of the year may not be always be coupled with anomalously high abundances in the second half of the year. Although data collection spanned a 9-year period, data from February 2013 to January 2014 was excluded from this analysis, as only limited sampling took place during that period.

The final model outputs provided (for each species) a mean daily number of individuals impinged for each month separately, along with lower and upper values that corresponded to 95 % confidence intervals.

5.5 Predicted annual mean, minimum and maximum losses by the SZC station

The predicted (unmitigated) losses that will be incurred by the SZC station were calculated by simply raising the mean, lower and upper estimates for SZB by the ratio of the two pumping capacities (i.e. 131.86/51.5 for each species).

5.6 Effect of the intake head design

The predicted effect of the proposed LVSE intake head design on the abstraction of organisms, was assessed by first multiplying all predicted values for SZC (unmitigated), by the factor of **0.383**. (Section 5.1.2)

5.7 FRR system mortality

The proposed drum screen mesh size for SZC is 10 mm allowing a direct comparison with the current mesh size employed at SZB. In the best practice guide for screening for intakes and outfalls Environment Agency (2005) recommend *"mesh size should be as small as is practical, and of no more than 6 mm aperture"*. However, Environment Agency (2010) acknowledge that at coastal sites a 6 mm mesh may lead to the risk of ctenophore blockage during summer months. Gelatinous ctenophores would more readily distort under drum screen conditions and squeeze through a 10 mm mesh screen (Environment Agency, 2010). After a consideration of clogging risk EDF Energy have proposed that SZC's fine filtration systems should have 10 mm mesh. The trash rack bar spacing might then be relaxed to 75 mm bar without exceeding design criteria for the drum screens.

5.7.1 Trash rack mortality

Located immediately before the drum and band screens, will be a series of trash racks, designed to protect the screens from debris but which will also prevent the passage of large fish. The racks, which will have a bar spacing of 75 mm can be raised for cleaning and any material that cannot pass through the bars will be sent to the debris recovery building (HCB). The debris recovery building has another trash rack with 200 mm bar spacing and fish that pass through this secondary trash rack will be returned to sea via the FRR tunnel. Any that remain (i.e. cannot pass through 200 mm) will go to waste. It is assumed (subject to review of the SZC design), that all organisms that cannot pass through the primary trash rack (75 mm bar spacing) will suffer 100 % mortality, even if they would then pass the secondary trash.

Each of the 24 key species has a different body shape and maximum size, and therefore the size at which an individual will be able to pass will depend on its species. The proportion of the total number of that species that will not pass will depend on its size distribution in the cooling water systems.

5.7.1.1 Calculation of annual length distributions

The final impingement dataset also included the number at length of the key species, by sampling visit. These were used to provide an annually-raised length distribution for each of the key species. The number of individuals in the annually raised length distribution equalled the number of individuals estimated to have been impinged by SZC per year.

Several steps were required to account for seasonal growth in the length distributions:

- First, the samples were grouped by month, and the numbers at length were summed. The total numbers at length were then divided by the number of samples in that month. This provided a standardised length distribution for each month separately.
- Next, the 12 monthly length distributions were summed, and their total numbers at length were divided by 12 to give a single standardised length distribution representing one year.

- Finally, the numbers at length were raised to an annual total by multiplying by (number of individuals impinged by SZC/number of individuals in the standardised length distribution).

(These annually raised length distributions were also used in the calculation of the EAVs - see BEEMS Technical Report TR383).

5.7.1.2 Calculation of the cut-off for passing through the 75 mm bar spacing.

For each species, its ability to pass through the 75 mm trash rack will depend on its width, which can be calculated from its length. However, HPB already has trash racks fitted with a 75 mm bar spacing, and length data show that for some species (e.g. cod and bass), even fish with a calculated width > 75 mm can pass through to the drum screens (BEEMS Technical Report TR456). This indicates that the passage of a fish through the trash racks may not simply be limited to its calculated width. This has been accounted for in calculations on the proportion of a species that will or will not pass the 75 mm bar spacing. Species were grouped depending on how their passage through the trash racks was assessed:

- Group 1. For some species (e.g. sand gobies), expert judgement was used to conclude that all individuals would pass, irrespective of size (i.e. proportion retained by the trash racks = 0).
- Group 2. For all other species, the width of the largest observed individual was calculated. Group 2 species are those whose calculated width of the largest observed individual was ≤ 75 mm (e.g. river lamprey *Lampetra fluviatilis*). It was assumed that no individuals will be retained by the trash racks.
- Group 3. For almost all the remaining species the calculated width of the largest SZB fish was ≥ 75 mm, indicating that a proportion of individuals will not pass. Most of these species were recorded during impingement sampling at HPB, which is already fitted with 75 mm trash racks. For this report, the length of the largest observed individual to pass through the 75 mm HPB trash rack was used as the maximum size that would also pass through the proposed 75 mm SZC trash rack. The proportion of a species that would be retained on the trash rack was calculated from the annually raised SZB length distributions of that species (Section 5.7.1.1).
- Group 4. For two species (sea trout and Allis shad), the calculated width of the largest SZB fish was ≥ 75 mm, indicating that a proportion of individuals will not pass through, but the species was not recorded at HPB. The approach used for Group 3 fish could not be used. In this case, the length of a fish with a width of 75 mm was calculated as the maximum size that would pass through the trash racks. Given that for many species, individuals of a calculated width were able to pass through the trash racks, this is likely a conservative approach.

For each species, the proportion of fish that would be retained by the 75 mm trash rack was applied to the number of fish that passed through the intake head, giving the number of fish lost to the trash racks (Table 5). Fish passing through the trash racks will go on to encounter the drum or band screens.

Table 5 Proportion of fish, by species that will not pass through the 75 mm wide trash racks, and the length size used for the cut-off

	Calculation type	Length of largest SZB fish (mm)	Calculated width of largest SZB fish (mm)	largest observed HPB fish	Calculated length at 75 mm width (cm)	Proportion not passing trash rack	Comment
Sprat	Group 2	260	53			0.000	All observed fish will pass
Herring	Group 3	445	94	224		0.715	Use largest observed HPB fish length (impingement datasets)
Whiting	Group 2	525	48			0.000	All observed fish will pass
Seabass	Group 3	850	232	657		0.0002	Use HPB largest observed individual (TR456)
Sand goby	Group 1	100	NA	74		0.000	All observed fish will pass
Dover sole	Group 3	440	124	449		0.000	Use HPB largest observed individual (TR456)
Dab	Group 3	380	160	139		0.419	Use largest observed HPB fish length (impingement datasets)
Anchovy	Group 1	230	NA	169		0.000	All observed fish will pass
Thin-lipped grey mullet	Group 3	425	91	524		0.000	Use largest observed HPB fish length (impingement datasets)
Flounder	Group 3	425	119	339		0.031	Use largest observed HPB fish length (impingement datasets)
Plaice	Group 3	355	150	382		0.000	Use HPB largest observed individual (TR456)
Smelt	Group 1	250	NA			0.000	All observed fish will pass
Cod	Group 3	895	133	709		0.006	Use HPB largest observed individual (TR456)
Thornback ray	Group 3	765	503	952		0.000	Use HPB largest observed individual (TR456)
River lamprey	Group 2	405	25			0.000	All observed fish will pass
European eel	Group 2	895	56			0.000	All observed fish will pass
Twaite shad	Group 3	490	103	299		0.883	Use largest observed HPB fish length (impingement datasets)
Horse mackerel	Group 2	355	71			0.000	All observed fish will pass
Mackerel	Group 2	405	69			0.000	All observed fish will pass
Tope	Group 2	625	63			0.000	All observed fish will pass
Sea trout	Group 4	530	121	not at HPB	33	1.000	Calculate length at 75 mm width
Allis shad	Group 4	610	128	not at HPB	36	1.000	Calculate length at 75 mm width
Sea lamprey	Group 2	795	50	807		0.000	all observed fish will pass

5.7.2 FRR survival

Well-designed FRR systems have been reported to achieve 80−100 % survival rates for robust species such as eel *Anguilla anguilla* , plaice *Pleuronectes platessa* and flounder *Platichthys flesus*, and moderate rates (~50−60%) for demersal species such as the robust gadoids (e.g. cod). However, survival rates for delicate pelagic species such as herring, sprat and shad (twaite shad *Allosa fallax* and Allis shad *A. alosa*) are usually low (<10%, Environment Agency, 2005). The proposed FRR system for SZC at both the band and drum screens has been designed to achieve high rates of survival for European eels (eel, *A. anguilla*) and lamprey (river lamprey and sea lamprey *Petromyzon marinus*), and it is expected that survival rates for other epibenthic (flatfish including rays) and demersal species will also be higher than achieved in older designs. For the purpose of this study the conservative FRR recovery rates given in the Environment Agency science report (2005) are used as the basis for FRR survival rates.

However, the estimates of FRR survival have been modified to account for the SZC location and station design. The SZC FRR system will discharge inside the Sizewell-Dunwich Bank and there is potential for a few fish discharged from the SZC FRR to be subsequently taken up by the SZB intake. Modelling confirms that this risk is negligible at the proposed SZC FRR outfall locations (BEEMS Technical Report TR333), but it is conservatively assumed that any such fish will suffer 100 % mortality during their passage through SZB. The SZC mortality is increased to compensate for this effect (Table 6).

As well as using drum screens for the main cooling water flow, SZC will be equipped with band screens to protect the essential and auxiliary cooling water systems. Due to their safety role, the band screens must be seismically qualified and capable of surviving an aircraft impact. The normal operating mode of such band screens is to be stationary and to only rotate intermittently at 6 hourly intervals unless significant clogging occurs. It is possible to fit an FRR system to the band screens, but this would have little to no purpose if the screens only rotated every 6 hours. It would, however, serve a purpose if the screens rotated continuously. The band screen manufacturer considers that the screens could be operated continuously at a ‘creep’ rotation speed of 0.5 metres per minute; any faster would have unacceptable implications for the operational life and maintenance of the safety-classified band screen motor and chains. For HPC, it was assumed that with the size of the band screens, at a rotation speed of 0.5 m min^{-1}, the fish retention time in the band screen fish buckets would be approximately 33 minutes at Mean Sea Level (MSL) and 50 minutes at the Lowest Astronomical Tide (LAT). As a conservative assumption, it was considered that demersal fish would not survive this time in the fish buckets. However, with a fish-friendly design ensuring they cannot fall out of the buckets during the predicted retention time, robust epibenthic species such as flatfish, eels and lamprey are expected to survive. The tidal range at Sizewell is lower than at Hinkley Point and the size of the SZC drum and band screens will therefore be smaller leading to reduced rotation time and reduced fish retention time in the fish buckets which will enhance survival. In this assessment we have assumed, as a conservative estimate, the same FRR survival estimates that were applied to the HPC band screens; i.e. that the fish survival percentages for epibenthic species will be the same for drum screen and band screen FRR systems, but that the survival of demersal species in the band screens will be 0 % (Table 6). This conservative assumption may be revisited when further information is available on detailed station design.

To calculate drum screen and band screen losses, the number of fish remaining after passage through the trash racks was apportioned into those that will encounter the drum screens and those that will encounter the band screens based on the proportion of water flowing through each (drum screens = 91 %; band screens = 9 % of the cooling water). This is considered a reasonable approach as the approach velocity is identical for all of the SZC drum and band screens. The proportion of numbers of fish that would be retained by the drum and band screens was calculated for each species separately.

5.7.3 Other factors which could potentially affect FRR survival rates

In principle two issues that have not yet been discussed in this report could affect the survivability of impinged fish in the SZC FRR system:

a. Overloading of the SZC FRR system by dead fish that do not survive the recovery process (mostly pelagic fish such as sprat and herring) causing fatal oxygen depletion in parts of the system where fish densities are the highest (the drum screen buckets)

b. Clogging of the SZC FRR system by ctenophore blooms in summer

The SZC seawater filtration system is designed to protect the CW condensers and heat exchangers from blockage from marine organisms. The system was designed based upon operational experience at EDF Energy coastal power stations without impingement mitigation technology and as such there was no assumption in the design of the filtration system for reductions in fish impingement due to the SZC Low Velocity Side Entry (LVSE) intake heads. The SZC filtration system has been designed to have considerable capacity to respond adaptively to extreme fish densities at the drum and band screens by increasing the rotation rate of the screens such that organisms are returned to sea via the FRR system at a faster rate. For example, the drum screen rotation rate can be increased from the normal 2.5 m min^{-1} to 10 and then 20m min^{-1} in response to different screen loadings i.e. the system can provide an 8-fold increase in filtration capacity under extreme conditions. The band screens are even more adaptable and can provide a 20-fold increase in filtration capacity by increasing the rotation rate from 100 minutes per rotation to 20 and then 5 minutes.

The main clogging risk at Sizewell are from winter sprat inundations and particularly from summer ctenophore blooms. As noted in Section 5.7, after consideration of the clogging risk from ctenophores, EDF Energy is proposing to to fit 10mm mesh filters at SZC. Operational experience at Sizewell B has shown that such a mesh size has avoided problems of clogging from either ctenophores or sprat. In terms of relative risk, SZC would be at much lower risk than SZB from sprat inundations due to the use of LVSE intake heads at SZC which it has been calculated will reduce sprat impingement per cumec by a factor of 0.38 (Section 5.1.2). A study for HPC (BEEMS Technical Report TR493) determined that there was a negligible risk to fish survival in the HPC FRR system due to dead impinged fish (overwhelmingly sprat). The rotation path lengths of the SZC drum and band screens will be smaller than those deployed at HPC due to the smaller tidal range at Sizewell and therefore fish residence times in the fish buckets will be shorter. The risk to fish survival from dead fish in the fish buckets is, therefore, expected to be lower at SZC than at HPC and therefore also negligible

Ctenophores are present for most of the year at Sizewell but only occur in dense blooms in summer (typically in June). The proposal to use a 10mm mesh means that the majority of entrapped ctenophores will be entrained rather than impinged. There have not been any shutdowns at Sizewell B which also uses 10mm mesh filtration due to gelatinous species. With negligible clogging risk there remains the question of whether the loading of ctenophores would reduce fish survivability in the SZC FRR system. As can be seen from Figure 3 the weight of fish impinged during summer ctenophore blooms is extremely small. Much smaller peaks in ctenophore abundance occur at other times of the year but the additional ctenophore biomass is much smaller than the peak fish biomass that the FRR system can handle with negligible risk to fish survival.

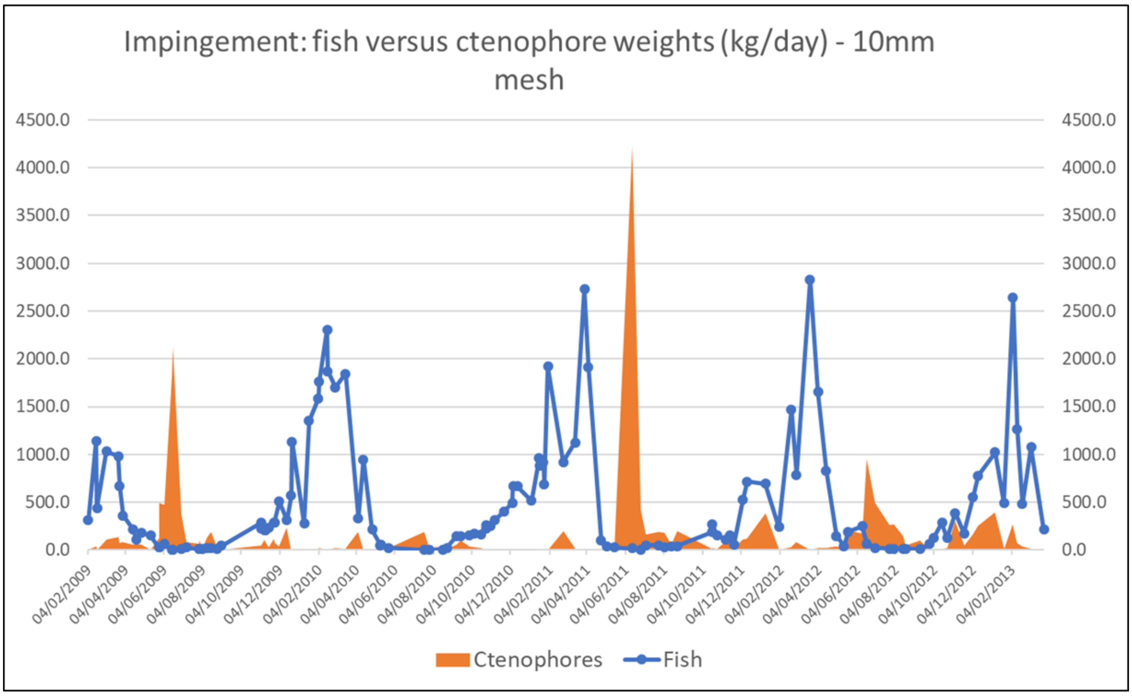


Figure 3 Comparison between estimated daily impingement weights for ctenophores and fish from SZB CIMP data.

Clogging of the SZC FRR system and the consequent risk to fish survival of recovered fish is, therefore, considered to be negligible.

5.7.4 Values used for FRR survival in the SZC impingement assessment

The predicted total mortality suffered by each species passing through the SZC cooling water systems, considering FRR survival, was calculated as:

Total mortality = Trash rack losses + band screen losses + drum screen losses

The results are shown in Table 6.

Table 6 Predicted FRR mortality by species through the SZC drum and band screens

	Proportion lost		
	Drum	Band	Species group
Sprat	1.000	1.000	pelagic
Herring	1.000	1.000	pelagic
Whiting	0.506	1.000	demersal
Bass	0.506	1.000	demersal
Sand goby	0.206	0.206	epibenthic
Sole	0.206	0.206	epibenthic
Dab	0.206	0.206	epibenthic
Anchovy	1.000	1.000	pelagic
Thin-lipped grey mullet	0.506	1.000	demersal
Flounder	0.206	0.206	epibenthic
Plaice	0.206	0.206	epibenthic
Smelt	1.000	1.000	pelagic
Cod	0.560	1.000	demersal
Thornback ray	0.206	0.206	epibenthic
River lamprey	0.206	0.206	epibenthic
Eel	0.206	0.206	epibenthic
Twaite shad	1.000	1.000	pelagic
Horse mackerel	1.000	1.000	pelagic
Mackerel	1.000	1.000	pelagic
Tope	0.206	0.206	epibenthic
Sea trout	0.506	1.000	demersal
Allis shad	1.000	1.000	pelagic
Sea lamprey	0.206	0.206	epibenthic
Salmon	0.506	1.000	demersal

5.8 EAV conversion factors

Equivalent Adult Values (EAVs) are used to adjust the number of lost juveniles to a corresponding number of lost adults. This adjustment is required because juveniles suffer higher natural mortality when compared with adults of the same species, and the loss of one juvenile does not result in the loss of one adult. Conversion of the numbers impinged to the equivalent number of adults is a simple matter of multiplying the former (the total impingement mortality) with the appropriate EAV value for each species. The EAV values used were calculated using a method developed as part of the BEEMS programme (BEEMS Technical Report TR383). The reliability of the EAV calculation method has been extensively evaluated for HPC studies in BEEMS Technical Report TR456 and the calculation method used at Sizewell is the same as that used for HPC, in particular the natural mortality corrections applied are the same. These calculations use data on the size distribution of the impinged fish along with information on size at age and size at maturity. Since the model outputs giving rise to the SZB estimates and SZC predictions were based on the combined 2009 – 2017 data, only a single EAV could be calculated for each species (Table 7) that used the size distribution from the whole sampling period (see Section 5.7.1.1 for details on the calculation of the size distribution).

All methods for calculating EAVs are subject to biological uncertainties. The method used for this assessment and described in BEEMS Technical Report TR383 is based upon the best available science at the time of writing and uses peer reviewed input parameters appropriate to the local populations at Sizewell.

A key advantage of this method is that the approach represents a method for calculating EAV that can be widely applied to many species and requires data that are more readily available than the method described by Turnpenny (1989) that has been used for some previous power station effects analyses. Turnpenny (1989) defined the EAV as “the average lifetime fecundity of an adult that has just reached maturity which is required to replace that juvenile”. To calculate the EAV curve or construct the associated life table for a given species, a variety of parameters are required, such as the average fecundity of each female age class, the number of age classes in the population and the average fecundity of mature females of the final age class. Many of these parameters are difficult to reliably estimate, even to orders of magnitude. Consequently, Turnpenny (1989) only provided EAV life tables/curves for six commercial species (cod, whiting, plaice, Dover sole, dab and herring) and the accuracy of these values is uncertain.

5.8.1 Discussion of the Spawner Production Foregone method for calculating EAVs

For HPC assessment purposes an alternative method for calculating EAVs has been suggested by stakeholders using the ‘Spawner Production Foregone’ (SPF) method of deriving equivalent adult value factors. The use of this method has been occasionally attempted in the past for assessing power station effects (predominantly in the USA). The SPF approach calculates the total loss to the population as the sum of equivalent adults lost from the spawning population in that year, plus the future spawning potential of lost fish that would have matured in previous years. In so doing the assumption is made that long lived fish could have spawned in multiple future years after they reached maturity and therefore some fish populations could have an EAV value of greater than 1, whereas the Cefas method has a maximum EAV value of 1. The major assumption of the SPF methodology is that long lived species would survive to spawn in multiple future years after reaching maturity (for short lived species the differences between the results produced by the SPF and Cefas EAV methodologies can be small dependent upon the impingement catch distribution). However, for many species this is an invalid assumption as after reaching maturity (often before reaching maturity), the fish become at risk of fishing mortality which is frequently much greater than natural mortality such that few fish live to spawn on multiple occasions; this effect is particularly pronounced for species such as cod. Since fishing mortality can and does occur before the age of maturity, the Cefas method described in BEEMS Technical Report TR383 for calculating EAVs can also overestimate the EAV for populations with large numbers of fish at or near maturity, however the potential error in ignoring fishing mortality is much less than with the SPF method.

The other significant issue with the SPF method is that ICES fisheries assessment models do not account for production foregone as they simply reflect the Spawning Stock Biomass (SSB) of fish alive at a particular point in time. So, unless the time dimension is kept for the SPF method (which combines several time periods), it is difficult to see how the population effects derived using SPF EAVs are comparable with ICES SSBs, particularly as the SPF method also ignores fishing mortality (something that ICES stock assessments fully account for). Simply put the TR383 method computes the value of a lost mature adult as 1 fish lost from the Spawning Stock Biomass, it does not try to account for the potential future production from that fish if it lived to spawn again. As such it is compatible with the methods used to assess the effects of fishing and the derivation of the spawning stock biomasses by ICES. Comparison with ICES stock assessments is the standard method of assessing the effect of fishing or power station entrapment and the ability to undertake reliable, quantitative comparisons of power station mortality against SSB is fundamental to the assessment of SZC effects. The use of the SPF method has, therefore, not been taken forward in this assessment.

5.8.2 Potential detail change to the EAV calculation methodology in TR383

The effect on EAV results if the calculation stopped at 50 % maturity and not the 100% maturity used in TR383 has been queried. The rationale for this question is that the Turnpenny (1989) method is based upon ages at 50% maturity as are some stock assessments, and a revised methodology could be more conservative.

It was considered that such a change probably wouldn't materially affect the predicted SZC losses but Cefas examined whether this initial view was justified. The potential effect of such a change was checked for three different types of fish taxa with different life cycle parameters (bass, cod, herring) and calculated EAVs increased by 3.6%, 11.5% and 4.3% resulting in consequential changes in predicted entrapment losses by the same percentage. For example, the EAV for bass would change from 0.224 to 0.232 resulting in an increase in unmitigated impingement losses from 1.32% SSB to 1.37% SSB and cod losses would increase from 0.0035% SSB to 0.0039% SSB. Such changes are considered immaterial to the SZC assessment and have not, therefore, been included in this assessment report.

5.9 Conversion from EAV numbers to equivalent weight

As most stock information (particularly for commercially-exploited species) is given as weight rather than number of fish, the EAV numbers of impingement losses were converted to EAV weights (t), using values of mean weight per species (Table 7). The preferred method of calculation would have been to calculate the mean weight of adults only for all species. However, the final method used depended on the data available for that species.

For some species where international stock assessments are carried out, the datasets contain the mean weight of fish in each age group, the catch numbers in that age group and the proportion of fish mature in each age group. In this case, the numbers of mature fish caught in each age group was calculated by multiplying the number caught in the age group by the proportion that were mature. Multiplication of the number mature in the age group by the mean weight of an individual in the age group provided the total weight of fish in the age group. The weight of all fish in the age groups was summed and then divided by the total number of mature fish to give a mean weight of each mature fish.

However, this level of data is not available for all species, and alternative methods were used for these species. For salmon and trout for example, the EA catch statistics reports (Table 8) give the average weight of individuals of each species caught in the fishing year, based on catch returns, and these values were used. For lamprey the mean weight of fish entering the Ouse was used, as it was assumed that these fish are entering the estuary to spawn, and are therefore mature.

In each case, judgement was based on the data available.

Table 7 EAV metrics and mean weight of individuals used to convert the numbers impinged to adult equivalent numbers and weights at SZC. See BEEMS Technical Report TR383 for full EAV calculations. Species where an EAV could not be calculated are highlighted in yellow and the impingement losses are overestimates

Species	EAV	Mean weight per individual (kg)	Data source for mean weight
Sprat	0.751	0.011	ICES catch weight at age data
Herring	0.715	0.189	ICES catch weight at age data
Whiting	0.356	0.286	ICES catch weight at age data
Seabass	0.224	1.531	ICES catch weight at age data
Sand goby	1.000	0.002	Mean size of gobies impinged at SZB
Sole	0.213	0.214	ICES catch weight at age data
Dab	0.445	0.041	ICES catch weight at age data
Anchovy	0.974	0.021	Mean weight of anchovy the Bay of Biscay catches
Thin-lipped grey mullet	0.083	0.520	Estimated from weight at 75 % maturity (age 3.5)
Flounder	0.462	0.082	Estimated from weight at 21 cm (fully mature - ICES)
Plaice	0.345	0.246	ICES catch weight at age data
Smelt	0.761	0.017	Estimated from length at maturity (knife-edge)
Cod	0.359	2.602	ICES catch weight at age data
Thornback ray	0.193	3.193	Mean length of market sampled fish in southern N. Sea

River lamprey	1.000	0.079	Mean weight of lampreys entering Ouse fishery in 2018 – regulator comment to Version 1 of this report
European eel	1.000	0.329	Mean of a mature female (568.9 g) and male (89.9 g) adult (Aprahamian, 1988), assuming a 50:50 sex ratio
Twaite shad	1.000	0.313	Calculated from adult size of 32 cm
Horse mackerel	1.000	0.140	Mean weight in the catch
Mackerel	1.000	0.319	ICES catch weight at age data
Tope	1.000	6.900	Modal length of fish in observer catches = 100 cm
Sea trout	1.000	1.734	from EA catch statistics
Allis shad	1.000	0.572	Calculated from adult size of 40 cm
Sea lamprey	1.000	1.212	Weight of single individual (length = 79 cm) impinged at SZB
Salmon	1.000	3.684	from EA catch statistics

Note: The use of an EAV of 1 results in an overestimate of impingement effects for most species e.g. it is known that the impinged eels at Sizewell were yellow eels that will suffer additional natural mortality before reaching the adult silver eel stage. Similarly, not all of the impinged twaite shad were adults.

5.10 Evaluating the effect of SZC impingement losses – comparison with ICES stock estimates

Fish mortality due to impingement at SZC can be considered as a form of fish harvesting.

Fish stocks in the Northeast Atlantic are managed partly through the EU Common Fisheries Policy (CFP), whose objective is to maintain or rebuild fish stocks to levels that can produce their maximum sustainable yield (MSY). The International Council for the Exploration of the Sea (ICES) advises public authorities with competence for marine management including the European Commission (EC). ICES was established in 1902 and its advice integrates the work of approximately 5,000 scientists from over 700 marine institutes in the organisation's 20 member countries and beyond.

ICES' advice is produced through a process which is set up to ensure that the advice is based on the best available science and data, is considered legitimate by both authorities and stakeholders and is relevant and operational in relation to the policy in question.

The basis for the advice is the compilation of relevant data and analysis by experts in the field, normally through an expert group which includes core researchers in the field. This analysis is peer reviewed by scientists who have not been involved in the expert group and have no direct interest in the matter. To support the stock-by-stock management system, ICES provides advice on fishing opportunities and stock status for individual stocks.

For many species, annual analytical assessments are carried out that utilise information on life history, fishing effort and catches to assess the size of the stock, in particular the spawning stock biomass (SSB). To undertake an assessment ICES scientists have to evaluate information on the life history and fishery characteristics of a stock to determine the most scientifically appropriate geographical area in which to assess the stock (the stock unit). Stock assessments are carried out by Expert Groups (also known as Working Groups), each of which is responsible for a specified number of species and stocks. The outcomes of the assessments are released as official ICES advice.

Wherever possible, in cases where a full analytical assessment is available for a species impinged at Sizewell, and the SSB has been estimated, the SZC predicted impingement losses were compared with the ICES estimated SSB for the stock area as these estimates provide the most robust peer reviewed scientific evidence. As the annually raised impingement numbers were based on data collected in 2009 – 2017, comparisons were made with the average SSB values over this time.

This is the preferred measure for determining the effects of fish impingement (and entrainment) losses and this is how the much larger environmental effects of fishing are internationally assessed and managed. It must be emphasised that the comparison with the SSB in the assessment year(s) conducted in this report are not a full fisheries population assessment and in stocks where the population biomass is heavily dependent upon new recruits which suffer a high rate of natural mortality this simple measure can provide a misleading overestimate of impingement effects. However, a full population assessment is both unnecessary and disproportionately difficult to undertake for species where impingement effects are negligible. If the predicted effects of impingement on a particular species were above the precautionary 1% negligible effects threshold used in this report (Section 6) a full population assessment is one of the steps that could be considered to reduce uncertainties and to determine if there was in fact any risk to the sustainability of the population.

5.10.1 Are ICES stock units appropriate for assessing the effects of SZC on fish populations?

To undertake their stock assessments ICES' scientists have identified biological stock areas that describe the distribution of a stock. These may be different from the areas defined by the EU, for example, for the management of fishing quotas and technical measures. Identification of appropriate stock boundaries has been a central theme of ICES' coordinated effort since its formation in 1902 and major advances in understanding have, and continue to be, made.

SZC stakeholders have queried the appropriateness of some of the existing stock units, particularly for bass which has one of the largest stock units of the key fish species included in the SZC effects assessment. In particular, they have queried whether the stock areas being used for the assessment of impacts to certain species consider the impact to local sub-populations given that several papers (including papers produced by ICES) provide evidence of sub-populations and more complex heterogeneous population structures.

There is extensive literature focused on specific aspects of fish migratory behaviour and this information is periodically reviewed by the appropriate ICES working groups. The questions then are do the specific behaviours described in the research papers change the weight of evidence used to assign stock units for management purposes? This section focuses on how ICES determines stock identity for fisheries management purposes.

As a result of decades of research, it is clear that the population structures of marine species fall along a continuum from panmictic (e.g. European eel, *Anguilla anguilla*) to numerous distinct sub-populations (e.g. North Sea herring, *Clupea harengus*) with the majority of species exhibiting complex structure. In the open sea, the sub-populations of many species mix to a considerable extent; especially during summer feeding and on nursery areas, with harvesting affecting multiple components of the overall population simultaneously. Spawning areas have been delineated for many fish species, but determination of the amount and timing of mixing is much more complex. Creation of appropriate stock units to manage harvesting has to consider all of these issues and it is recognised that in so doing harvesting may not always be optimal.

The current ICES stock boundaries are the result of decades of research and are inevitably compromises that try to embrace known uncertainties. Stock unit boundaries are not changed without careful weighing of the evidence for the need and the value to be gained from a change. However, stock boundaries are subject to periodic review and do change as a result of acquired evidence; e,g. North Sea sandeel where understanding of the vulnerability of largely sedentary post-settlement sub-populations led to a reorganisation of the original four stock units into seven units; with some stock units becoming larger and some smaller. For some species it is known that there is strong site fidelity during parts of the species lifecycle and/or seasonally, and some researchers cite this evidence to justify potential changes to stock units. However, it can be the case that there is considerable mixing of the sub-populations and an appraisal of all of the evidence leads to a conclusion that there is no evidence that a change could lead to better assessment and management advice. In such cases, the risks of realignment of stock boundaries are not warranted.

North Sea herring is one such example where the population is known to consist of several different sub-populations that have discrete well identified spawning areas but the population is managed as a single stock because of the degree of mixing that takes place. However, despite the use of a single stock unit, when the population collapsed in the 1970s the most at-risk spawning area was subsequently protected by limiting fishing effort in that location during the spawning season; i.e. adaptive management measures can be, and are, used to protect sub-stocks even within a single stock unit is the basis for ICES' advice.

Bass is a species where appropriateness of ICES stock units has been queried, so management of this species is specifically described in Section 5.10.2.

5.10.2 Scientific status of the ICES Bass stock unit and the consequent SSB estimate

The ICES stock unit for bass is spatially very large and during consultation with stakeholders the accuracy of this spatial extent has been queried. This section provides a summary of the latest ICES' advice and the aim of current scientific studies.

"Based upon the most up to date science, ICES considers the assessment stock unit for bass as Divisions 4b-c, 7a and 7d-h (central and southern North Sea, Irish Sea, English Channel, Bristol Channel, and Celtic Sea, Figure 4*). Previously, Pawson et al 2007 recommended amended stock units for assessment purposes based upon some UK tagging studies. However, scientific knowledge about bass has advanced since these 2000-2005 studies and ICES continues to recognise the 4b-c, 7a and 7d-h stock unit as the most appropriate for bass stock assessment purposes based upon all of the available scientific evidence.*

ICES recognises that there are probably separate bass sub populations based upon preferred residence areas at certain life stages and seasons in different areas (e.g. North Sea, Irish Sea) but studies have shown that because of a high degree of intermixing these 'sub populations' have the same stock dynamics which would not be the case if the fish were from separate stock units. The stock identity for bass is under regular review and was last assessed by ICES working group scientists in 2018 when it was concluded that the current stock unit continued to be appropriate for assessing the sustainability of the bass stock.

Scientific studies on bass stock identity including tagging programs, microchemistry and genetics are currently underway that are designed to provide more information on the movements of sea bass and the levels of mixing between stocks. The primary purpose of these studies is to determine whether the Bay of Biscay stock unit is indeed a separate stock or whether it should form part of the 4b-c, 7a and 7d-h stock". (pers. comm. Lisa Readdy Cefas, UK member of ICES Working Group for the Celtic Seas Ecoregion (WGCSE), 12 February 2020. This working group is responsible for the assessment of sea bass in Divisions 4b-c, 7.a and 7.d-h).

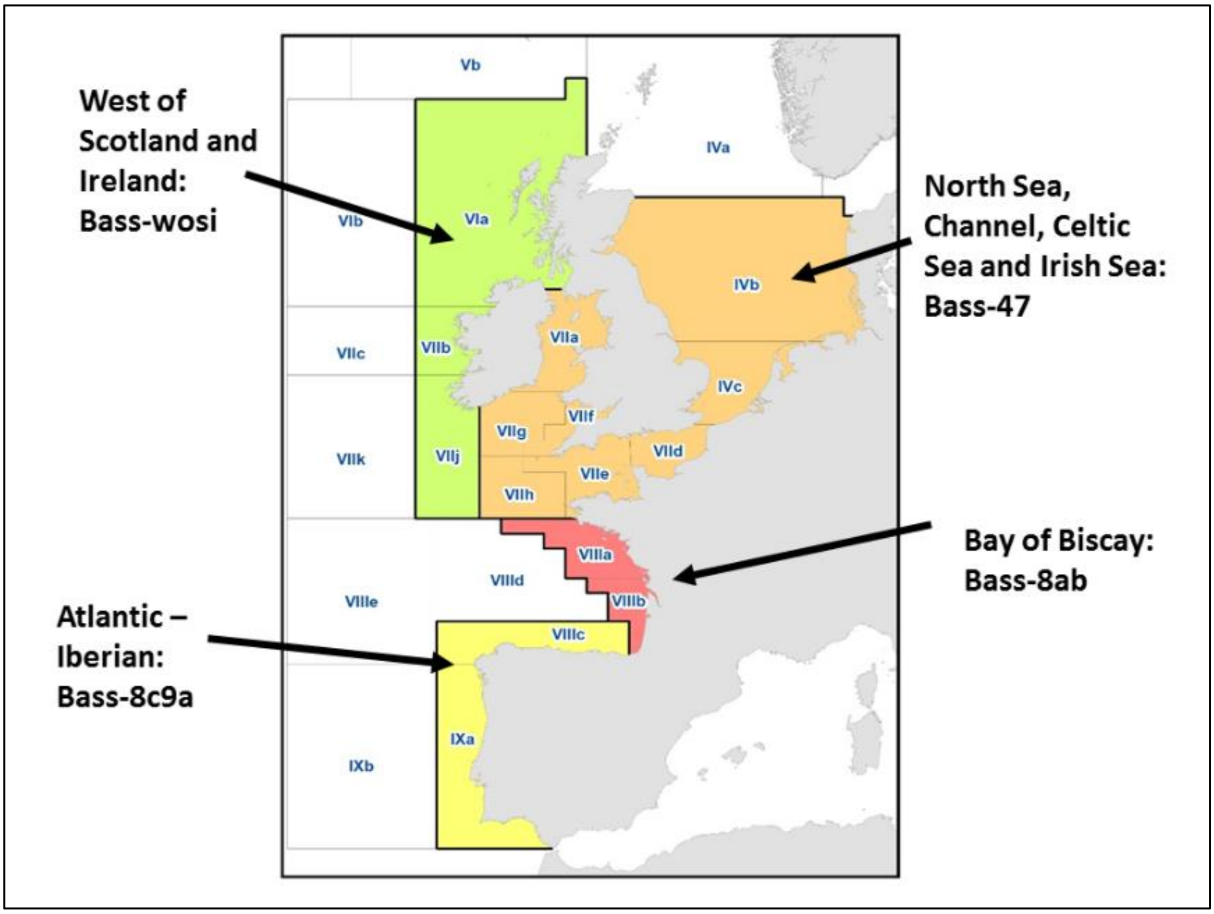


Figure 4 ICES stock units for bass (ICES WGCSE 2019)

In addition to questions about the separateness of the Bay of Biscay bass stock unit, there are similar questions about whether the bass in Irish coastal waters are indeed functionally separate (ICES WGCSE 2019). Stock identity studies are therefore currently focussed on the whether the stock unit size for bass should be expanded and there are no suggestions within ICES of a reduction in the bass stock unit relevant to the Sizewell C assessments.

Finally, after overfishing was identified on the 4.b-c, 7.a and 7.d-h stock bass stock ICES recommended a series of technical measures to reduce fishing mortality and to recover the size of the bass spawning stock biomass. These were implemented under the EU Common Fisheries Policy and the evidence to date is that the stock has partially recovered as predicted by the stock dynamics modelling that uses the 4.b-c, 7.a and 7.d-h stock identity.

ICES recommendations are science led and use the most up to date evidence available. ICES have recently reviewed the bass stock identities and found no compelling evidence to change the existing definitions. We therefore conclude there is strong confidence in the validity of the ICES' stock unit used as the basis for the SZC effects assessment on bass.

5.10.3 Conclusions on the validity of ICES stock units

Given the extensive body of ongoing research that supports the ICES' stock definitions, the questions that must be asked are whether it is appropriate, or proportionate, for EDF Energy to attempt to derive new stock assessment unit boundaries for the assessment of the effects of SZC and would any such new boundaries have any scientific credibility? Specifically, would such an effort add anything to advance the sustainability of fish populations given that the predicted effects of SZC (Table 18) are orders of magnitude below those from commercial fishing which is managed using ICES' stock boundaries? We have concluded that ICES' stock boundaries are compromises but they are based on a mature weighing of the best scientific evidence available and they are relied upon by governments to manage fish populations in the waters of all EU member states. Given the negligible predicted SZC impacts compared to those of fishing, and the precautionary nature of ICES' estimates of SSBs, we can find no justification not to use the ICES' stock definitions to assess SZC effects on fish.

5.10.4 How does ICES deal with non-fishing impacts on stocks?

In general, the ICES' approach to advice on fishing opportunities integrates ecosystem-based management with the objective of achieving maximum sustainable yield (MSY). Many of the models used by ICES to estimate MSY and associated parameters assume that factors not explicitly included in the models either remain constant or vary around an historical long-term average. Marine ecosystems are dynamic and fish stocks *may* change not only in response to the fisheries but in response to anthropogenic impacts and naturogenic effects as a consequence of bird and seal predation.

However, ICES does not include anthropogenic losses due to power stations in its stock assessments of commercially important species as such losses are considered as *de minimis*. This is supported by the predicted losses from SZC which show that the power station losses on fish populations would typically amount to less than 0.1% SSB or recorded international landings. (Pers. comm. Dr. C. O'Brien, Chief fisheries science advisor to Defra and vice President of ICES,18 February 2020).

5.11 Evaluating the effect of SZC impingement losses – Comparison with international landings data

For some species, although ICES collates all available fishery information, there are not enough data to carry out a full analytical assessment. While ICES may assess the status of the stock based on trends (e.g. trends in established surveys) and provide relative estimates of SSB, absolute SSB may not be estimated. In this case, we have compared impingement losses with the international landings for the stock area. However, such a comparison is unrealistically conservative as landings will be less than the stock size. For an unexploited stock, landings will typically be much less than 20% of the adult stock size and even for a heavily exploited stock, landings will rarely exceed 50% of the stock size.

5.12 Evaluating the effect of SZC impingement losses – Other comparative data sources

For other species alternative sources for population sizes, catches or landings were used:

a. Losses of thin-lipped grey mullet *Liza ramada* and were compared against ICES' Official Nominal Catches 2006 – 2017 (ICES, 2019), downloaded from the ICES website.

b. Environment Agency and Defra data were used to source catch data for salmon, sea trout, and eels (Table 8).

c. The silver eel SSB used for assessment purposes was from the Anglian RBD only.

d. For river lampreys, losses were compared against a spawning run size estimate for the Humber catchment made in 2018 by the Hull International Fisheries Institute (supplied by Dr J Masters, Environment Agency).

e. Data on shad are more difficult to source, as they are not part of a directed fishery and they may not be landed commercially in the UK. Some landings, of bycatch, are available in the ICES catch data, but these will substantially underestimate the size of the parent stocks. The twaite shad caught at Sizewell are considered to be part of a wider North Sea population that spawns in the rivers of Europe (predominantly in the Elbe but also in the Weser and the Scheldt) (BEEMS Scientific Position Paper SPP100), and the numbers of individuals impinged at SZC have been compared with abundance estimates from monitoring surveys conducted on the Elbe and the Scheldt.

f. Similarly, some landings data for smelt (*Osmerus eperlanus)* are available, but fishing in the UK for this species is limited by the number of fishing authorisations granted. The smelt caught at Sizewell are considered to be part of a wider North Sea population that spawns in at least the Rivers Elbe and Scheldt (BEEMS Scientific Position Paper SPP100). Numbers of individuals impinged at SZC have been compared with adult abundance estimates from monitoring surveys conducted in the River Elbe alone. The adult smelt stock has therefore been underestimated and the effect of SZC impingement has been overstated in this assessment. An assessment has also been made using relevant UK landings data as the existence of a southern North Sea population has yet to be proven.

g. Sand gobies are an unexploited fish stock and accordingly few abundance data are available. In Cefas Young Fish Surveys of the east and south coasts of England, gobies were the dominant species throughout the survey area, with highest densities recorded in the area from Flamborough to Winterton (region 1), followed by the area between Winterton and North Foreland (region 2), and the lowest densities between North Foreland and Portland Bill (region 3) (Rogers and Millner, 1996). For region 2, estimated densities in September of each year were approximately 41 individuals/1000 m^2, which is comparable with the abundances observed in the June BEEMS offshore survey (BEEMS Technical Report TR345). Population estimates for *Pomatoschistus* spp. in region 2 ranged between 36 – 197 million individuals between 1973 and 1995 (mean = 94.7 million, st. dev = 41.3 million individuals). However, studies on the catching efficiency of 2 m beam trawls showed that the gear is only 46% efficient at catching *Pomatoschistus* spp. over coarse sand (Reiss et al., 2006). Taking the trawl efficiency data, this would suggest that only 46 % of the gobies present in the areas surveyed during the YFS were recorded, leading to an underestimation of their abundance. This would imply that the mean population abundance of 97.4 million individuals should be raised to 205.8 million individuals to account for the *Pomatoschistus* spp. not caught.

A summary of the sources for SSBs, landings, catches or population and an indication of whether analytical assessments exist for each species is given in Table 8.

Table 8 Data sources used to provide information on relevant stock unit, landings and SSB

Species	ICES Working Group	Stock unit	Assessment type	Impingement effect comparator	Reference
Sprat	HAWG	Subarea 4 (North Sea)	Analytical assessment	SSB	ICES, 2018
Herring	HAWG	Subarea 4 & Divisions 3.a & 7.d (North Sea, Skagerrak & Kattegat, Eastern Channel)	Analytical assessment	SSB	ICES, 2018
Whiting	WGNSSK	Subarea 4, Division 7.d (North Sea, Eastern Channel)	Analytical assessment	SSB	ICES, 2018b
Bass	WGCSE	Divisions 4.b-c, 7.a, & 7.d-h (Central & southern N Sea, Irish Sea, English Channel, Bristol Channel & Celtic Sea)	Analytical assessment	SSB	ICES, 2018c
Sand goby	-	Not defined	Not assessed	Population abundance	Rogers and Millner (1996)
Sole	WGNSSK	Subarea 4 (North Sea)	Analytical assessment	SSB	ICES, 2018b
Dab	WGNSSK	Subarea 4 & Division 3.a (North Sea, Skagerrak & Kattegat)	Trends only	Landings	ICES, 2018b
Anchovy	WGHANSA	Given as 'Northerly anchovy'	Not assessed	Landings	ICES, 2018d
Thin-lipped grey mullet	-	Not defined	Not assessed	ICES Landings	ICES, 2019
Flounder	WGNSSK	Subarea 4 & 3.a (North Sea & Skagerrak and Kattegat)	Trends only	Landings	ICES, 2018b
Plaice	WGNSSK	Subarea 4 IV & Subdivision 20 (North Sea & Skagerrak)	Analytical assessment	SSB	ICES, 2018b
Cucumber Smelt	-	Not defined but includes the East Anglian coast and rivers on the European coast from the Elbe to the Scheldt	Estimated adult numbers migrating up river	Elbe populations EA landings & ICES Landings	BEEMS Scientific Position Paper SPP100 EA, 2018, 2017a, 2017b, 2015, 2014, 2013a, 2013b, 2013c; ICES, 2019
Cod	WGNSSK	Subarea 4 & Subdivisions 7.d & 20 (North Sea, Eastern Channel, Skagerrak & Kattegat)	Analytical assessment	SSB	ICES, 2018b
Thornback ray	WGEF	Subarea 4 & Divisions 3.a & 7.d (North Sea, Skagerrak, Kattegat & eastern English Channel)	Trends only	Landings	ICES, 2018e
River lamprey	-	Humber catchment	Estimated run numbers	Numbers converted to weight	EA, 2018, 2017a, 2017b, 2015, 2014, 2013a, 2013b, 2013c
Eel	WGEEL	Anglian River Basin District (RBD)	Biomass estimated	Estimated silver eel biomass	(Defra, 2018, 2015)

Species	ICES Working Group	Stock unit	Assessment type	Impingement effect comparator	Reference
Twaite shad	-	Not defined but includes the River Elbe and Belgian river Scheldt. A separate spawning population on the river Weser has not been included in the assessment.	Estimated adult numbers migrating up river	European populations in the Elbe. ICES Landings	BEEMS Scientific Position Paper SPP100 ICES, 2019
Horse mackerel	WGWIDE	Divisions 3.a, 4.b,c & 7.d (North Sea)	Trends only	Landings	ICES, 2018f
Mackerel	WGWIDE	Subareas 1–8 and 14, & Division 9.a (the Northeast Atlantic & adjacent waters)	Analytical assessment	SSB	ICES, 2018f
Tope	WGEF	North east Atlantic	Not assessed	Landings	ICES, 2018e
Sea trout	-	Not defined	Assessment based on CPUE	EA Catch numbers, UK	EA, 2018, 2017a, 2017b, 2015, 2014, 2013a, 2013b, 2013c
Allis shad	-	Garonne	Analytical assessment	Adult stock in 2009, ICES Landings	BEEMS Scientific Position Paper SPP071/s) ICES, 2019
Sea lamprey	-	Not defined	Not assessed	-	
Salmon	WGNAS	North Atlantic	North Atlantic	EA Catch numbers, UK	EA, 2018, 2017a, 2017b, 2015, 2014, 2013a, 2013b, 2013c

Working group acronyms:
HAWG - Herring Assessment Working Group for the Area South of 62°N
WGNSSK - Working Group on the Assessment of Demersal Stocks in the North Sea and Skagerrak
WGCSE - Working Group on Celtic Seas Ecoregion
WGHANSA - Working Group on Southern Horse Mackerel, Anchovy and Sardine
WGEF - Working Group on Elasmobranch Fishes
WGEEL - Joint EIFAAC/ICES/GFCM Working Group on Eels
WGWIDE - Working Group on Widely Distributed Stocks
WGNAS - Working Group on North Atlantic Salmon

6 Assessment of the significance of impingement effects

There are no formal UK regulatory guidelines for assessing the significance of fish mortality levels caused by impingement in coastal power stations and therefore any assessment must be based on expert judgment.

For the purposes of this assessment we have adopted two screening thresholds that have been selected such that impingement losses lower than the appropriate threshold will have negligible effects on the year to year sustainability of a fish population. Effects above the appropriate threshold would not necessarily indicate a significant adverse effect but require further investigation to determine whether significant effects were, in fact, present.

The thresholds have been selected based upon internationally accepted scientific practice for the sustainability of fish stocks under anthropogenic pressures:

c. For commercially exploited stocks and conservation species (which includes stocks that are not currently exploited): 1% of the SSB or, as a highly conservative proxy, 1% of international landings of the stock.
d. For unexploited stocks: 10% of the SSB or, as a highly conservative proxy, 10% of international landings of the stock.

In this section, scientific rationale for the selection of these screening thresholds are detailed.

Note that at the time of the HPC DCO the screening test that was applied and accepted for potentially significant environmental effects in the HPC Environmental Statement, shadow HRA and WFD was whether the predicted impingement of any of the assessed species was >1% of the SSB or fishery landings.

6.1 Screening thresholds for negligible effects in context

6.1.1 What is meant by a sustainable fish population?

Fishing is the selective removal (or harvesting) of fish. Impingement is therefore a form of fishing but of lower selectivity and much lower impact magnitude than fishing. Fish populations grow and replace themselves and they are therefore renewable resources. In the absence of harvesting, the population size of a stock does not increase indefinitely and stabilises around a maximum that a given habitat can support (the carrying capacity); i.e. it is under density control. The scientific basis for the sustainable use of a renewable marine resource evolved during the first half of the 20th century and is based upon a fundamental ecological principle of density dependent population regulation. As the abundance of a density regulated population is reduced by harvesting, per capita net production increases (by means of increased rates of growth, survival and reproduction), until the population cannot compensate for additional mortality after which point the productivity of the stock decreases and eventually becomes at risk of collapse. The production generated by this compensation (known as surplus production) can be harvested on a sustainable basis on a year on year basis (Rosenberg et al., 1993). Sustainability can therefore be framed as ensuring a sustainable harvest rate; i.e. where the rate of abstraction is less than or equal to the rate at which the population can regenerate itself. Determination of that rate for different fish stocks has been an internationally coordinated endeavour for more than 70 years and has led to well established stock assessment principles.

For well monitored stocks (data-rich stocks) quantitative stock assessment can be carried out which produces spawning stock biomass reference points below which a stock is either at risk of becoming unsustainable or is in an unsustainable condition, together with limits on the maximum harvest rate.

However, fisheries scientists are frequently required to advise on harvesting rates (also known as exploitation rates) of many data-limited stocks where an alternative precautionary approach is required. Several analytical approaches are applied in such circumstances which are largely determined by the availability and quality of the data. The different approaches are essentially based upon:

a. Limiting fishing mortality (F) to no greater than the natural mortality (M) of the species (determined by the life-history of the species).

b. For stocks where there is a record of fish landings, limiting F to the average fishing mortality (or index of fishing mortality) that did not lead to stock decline.

(Source: Food and Oceans Canada 2001)

[Note: The harvesting rate as a percentage of SSB is given by: $1^{-}e^{-F}$ (where F is fishing mortality) and for many demersal and benthic species in UK latitudes the adult M is in the range of 0.1 to 0.2]

Approach a. above is an internationally adopted management approach:

> *'Escapement strategies are used to manage short-lived species and exploitation rates of up to 20% are advised by ICES'* (pers. comm. Chief Fisheries Science Advisor to Defra, 2018).
>
> *'Limiting the exploitation rate to 10-20% of the estimated spawning stock biomass will ensure that fishing does not cause the stock to decline to unsustainable levels'* (Giannini et al., 2010).
>
> *'…a constant harvest rate of 20% of the spawning population became coastwide management policy …'*. (Hall et al., 1988).

M ranges from approximately 0.1 for some benthic species to >0.5 for some pelagic species at Sizewell (BEEMS Technical Report TR383); i.e. sustainable harvest rates vary with the lowest values being for long-lived, late maturating species. The sustainable harvest rate calculated from approach a. above approximates to but is more precautionary than the maximum sustainable yield and as such is well above the biological reference point where the stock would be at risk of becoming unsustainable. For many unexploited species that occur at UK latitudes, this approach implies a sustainable harvesting rate of 10%-20% for such species. However, this formula is not conservative for very short lived (or tropical species) where the sustainable value of fishing mortality is less than M, typically 0.25 to 0.5 M (Caddy and Csirke, 1983). Gobies are such a short-lived species reproducing within the first year of life. They are a very abundant species that is ubiquitous in European coastal areas to at least a depth of 20 m. The species produces pelagic larvae which are dispersed by tidal currents resulting in a lack of genetic diversity over large geographic areas. Sand gobies have an estimated M of 3.3 (Fishbase) implying a sustainable harvesting rate of greater than 50%.

On a precautionary basis a harvesting rate threshold of 10% SSB is considered appropriate as a screening threshold for potentially significant effects that may affect the sustainability of an unexploited fish stock.

6.1.2 Natural variability of fish stocks

Fish stocks are subject to considerable annual variability due to highly variable levels of recruitment, food availability and predation pressure. Individual populations and ecosystems are resilient to such high levels of variability. Impingement at SZB mirrors the variability of local fish populations as the power station is an efficient sampler with low interspecies bias unlike trawl or other net sampling techniques. As explained in Section 5.3, the discontinuous nature of the SZB impingement dataset meant that it was not possible to produce an estimate of year to year variability directly from data collected on site and instead a statistical model had to be fitted to the data.to derive impingement estimates (Section 5.4).

Some examples of the modelled year to year variability in local fish populations in the period 2009 - 2017 are shown in Table 9. The predicted variability for many species is substantially less than at the Hinkley Point estuarine location (BEEMS Technical Report TR456). This is as expected due to the composition of the fish assemblage at the Sizewell coastal location with substantially smaller numbers of 0 group fish whose numbers show the greatest year to year variability due to changes in annual recruitment.

Given the magnitude of such changes, with a minimum change of 130% and a maximum of 770% in year to year numbers, a <1% change due to impingement is negligible, particularly to predator-prey relationships which are adapted to cope with the much greater natural variability.

Table 9 Modelled year to year variations in SZB impingement numbers (2009-2017)

Species	The largest year-year changes in annual numbers from the Sizewell B CIMP dataset 2009-2017 (shown as the ratio of predicted impingement numbers in adjacent years)
Sprat	1.3
Herring	2.4
Whiting	2.8
Bass	1.9
Sand goby	5.6
Sole	2.1
Dab	7.7
Anchovy	4.1
Thin-lipped grey mullet	5.5
Flounder	1.6
Plaice	1.3
Smelt	1.9
Cod	2.2

6.1.3 Comparison with sustainable levels of harvesting rate for data rich stocks

In Section 6.1.1 the internationally accepted precautionary harvest rate of 10-20% SSB was described for unexploited species where little monitoring data exists. It is useful to consider a 1% negligible effects threshold in the context of sustainable harvest rates for data rich stocks which in many cases are much greater than 20% (see Table 10).

ICES produces estimates of the precautionary levels of fishing mortality beyond which sustainability is at risk (F_{pa}). Examples from ICES stock assessments are shown in Table 10. Set against such numbers, an impingement mortality of less than 1% from SZC is negligible. An additional 1% mortality in addition to the effects of fishing is in the noise for practical stock assessments and in practice such a level of effect is much smaller than that due to the uncertainty in the input parameters which are already assessed on a precautionary basis in the stock assessment.

Table 10 Sustainable fishing mortality values based upon a precautionary management approach for species relevant to Sizewell

Species	Sustainable fishing mortality reference values using precautionary approach (Fpa)	ICES Working Group Report	Coefficient of Variation of the SSB 1998-2017
Herring†	26%	HAWG	18%
Whiting‡	28%	WGNSSK	12%
Bass*	19%	WGCSE	23%
Sole‡	36%	WGNSSK	30%
Plaice‡	31%	WGNSSK	58%

† ICES, 2018a; ‡ ICES, 2018b; * ICES, 2018c

This point is further underlined by the actual predictions of total SZC entrapment losses in Table 18 of 0.01%, 0.03%, 0.03%, 0.00% and 0.00% of SSB for herring, whiting, bass, sole and plaice respectively.

6.1.4 An example of where screening thresholds for fish mortality have been applied for major infrastructure projects

A 10% screening threshold has been previously adopted by the Thames Tideway Strategy Group. This group comprised representatives from the Environment Agency, Port of London Authority, Thames Water and others and developed water quality standards for the regulation of dissolved oxygen levels in the Thames Tideway to protect fish from mortality associated with storm discharges through combined sewer outfalls (Turnpenny et al., 2004). The efficacy of different standards was compared using an ecotoxicological model, the Tideway Fish Risk Model (TFRM). The Turnpenny report argued that commercial fishery exploitation rates could be sustainable at >50% SSB, depending on the population dynamics of the species. Based upon the Turnpenny report, the TFRM considered annual mortality rates of up to 10 % to be sustainable for all species not subject to fishing mortality (i.e. the integrity of the population would not be threatened), and up to 30 % for longer-lived species such as seabass and salmon. The 10% value was also considered to be the practical minimum change likely to be detectable through ongoing routine WFD Transitional and Coastal (TrAC) water fish surveys.

The subsequent DCO application for the Thames Tideway Tunnel contained a review of the robustness of assumptions made in the TFRM including the definition of fisheries sustainability used in the model. The results of an independent expert peer review of the fisheries work were also provided (Thames Tideway Tunnel, 2013). The review conclusions were that the TFRM remained fit for purpose.

6.1.5 The appropriateness of a 1% SSB screening threshold for impingement effects

To have a negligible impact on a fish stock the predicted total anthropogenic harvest rate must be less than the value whereby the stock can replace itself on a year to year basis. For unexploited data poor species, a precautionary level of 10%-20% SSB is considered sustainable in international fisheries management practice. ICES advises in the context of current management policy which is to manage all species within sustainable limits by 2020; and policy measures have been recommended to the European Commission, which is responsible for managing marine fisheries in Europe, and are now being implemented in order to meet this objective as soon as possible in relation to the 2020 target.

For species which are heavily exploited by fishing a lower effect threshold for impingement is considered appropriate and 1% negligible effect screening threshold for annual impingement for all species provides a precautionary level which is negligible compared with fishing mortality on exploited stocks and would have no effect on their sustainability. A precautionary level of 1% is much less than the natural variability of any species at Sizewell which the ecosystem is adapted to and hence would have no significant effects on predator prey relationships. The use of a negligible effect threshold of 1% of SSB is, therefore, considered to be precautionary. At the request of stakeholders, the sustainability of this threshold and how it has been applied in the SZC assessment for the critically endangered European Eel stock is discussed in 6.1.6.

For non-exploited stocks a 1% threshold is highly precautionary based on fish population dynamics and any observed decline in stock numbers would be due to other factors well beyond the influence of SZC impingement. For such species a 10% screening threshold has been adopted in this assessment (e.g. for sand goby and thin lipped grey mullet)

6.1.6 Appropriateness of a 1% SSB threshold for the endangered European eel

Recruitment indices for glass eel arriving in continental waters decreased continually from 1980 to a low point in 2011. In 2011 a change occurred, and the recruitment trend has been increasing in the period 2011–2019 with a rate significantly different from zero. The reasons for the decline are uncertain but may include overexploitation, pollution, non-native parasites, diseases, migratory barriers and other habitat loss, mortality during passage through turbines or pumps, and/or oceanic factors affecting migrations. (WGEEL 2019). Whether the continental stock has declined as much as recruitment is unclear but the decline in recruitment was preceded by a decline in landings two or more decades earlier, indicating a decline of the continental stock. (Dekker 2003).

The reasons for the increasing recruitment trend are also unclear but may be linked to a substantial reduction in fishing mortality, some of which has been driven by economics, some by regulatory restriction. Commercial landings of glass eels have reduced by 97% from 2000 t in 1980 to less than 60 t in 2019. Commercial landings of yellow and silver eels have fallen by nearly 90% from 20,000 t in the 1950s to less than 2,700 t currently. Recreational fisheries have also reduced substantially in recent years although slope of the trend is unclear due to under reporting. (WGEEL 2019).

The International Union for the Conservation of Nature (IUCN) has assessed the European eel as ‘critically endangered’ and included it on its Red List in 2009. It renewed this listing in 2014 but recognised that: “*if the recently observed increase in recruitment continues, management actions relating to anthropogenic threats prove effective, and/or there are positive effects of natural influences on the various life stages of this species, a listing of Endangered would be achievable”* and therefore *“strongly recommend an update of the status in five years*.”

To date it has not been possible to determine the size of the eel stock SSB.

The current goal of the European eel management, set by the Eel Regulation (Council Regulation 1100/2007), is to “*reduce anthropogenic mortalities so as to permit with high probability the escapement to the sea of at least 40% of the silver eel biomass relative to the best estimate of escapement that would have existed if no anthropogenic influences had impacted the stock…. (The Eel Management Plans)… shall be prepared with the purpose of achieving this objective in the long term*.”

There are insufficient data to reliably determine the biological reference points used in ICES quantitative stock assessments and the purpose of EMPs is to gradually reduce total anthropogenic mortality to achieve a recovery in the adult stock. Given the 10 to 25 y+ taken for glass eels to reach maturation this will inevitably be a lengthy process. It is not clear that the recent (9 y long) trend in increasing recruitment is the early sign of a recovery or not.

For such a population it is reasonable to ask whether 1% of SSB represents a precautionary no effects threshold for SZC. There is no SSB estimate for the entire stock so the alternative measure would be 1% of landings (of 2700 t) or 27 t. Given the lack of biological reference points and the uncertainty surrounding eel stock dynamics, in this assessment it was not considered that 27 t would be sufficiently precautionary and instead the stance was taken of assessing the station effects against the Anglian RBD silver eel biomass of 78 t. with a 1% of SSB = 0.8 t (section 7.6.4.2). The Anglian RBD silver production is a small percentage of the entire stock SSB (on a crude measure of ratio of commercial landings it may only be 0.5% (13.9 t/ 2700

t) of the entire SSB and therefore 1% of the RBD SSB is a highly precautionary measure that could equate to approximately 0.005% SSB.

In conclusion, the 1% of SSB was retained for eel assessment but to be precautionary the SSB of just the Anglian RBD was used instead of population measures for the entire stock.

7 Impingement predictions for SZC - finfish

7.1 Predicted impingement without embedded mitigation measures

Annual estimates of all species impinged at SZB, and predictions for SZC are given in Appendix B. These are unmitigated values, with no adjustment for the embedded station mitigations or conversion to EAV equivalents. Ninety-one fish and 62 invertebrate taxa were recorded in 2009 – 2017. For fish, the most abundant nine species were sprat, herring, whiting, seabass, sand goby, sole *Solea solea*, dab, anchovy *Engraulis encrasicolus* and thin-lipped grey mullet and these contributed 95 % by number of all impinged fish. For invertebrates, the top six species were ctenophores, brown shrimp, pink shrimp, common prawn, swimming crab *Liocarcinus holsatus* and shrimp *Crangon allmani*. These six species contributed 98.9 % of the total invertebrate abundance, but it should be noted that ctenophores alone contributed 83.7 %.

The predicted unmitigated SZC impingement effects for the 24 key species after adjusting to equivalent adults are given in Table 11. Of the 24 key finfish species only the following 3 species exceeded 1%:

- Seabass (1.3 % of SSB)
- Thin-lipped grey mullet (2.5 % of landings) - as an unexploited stock, a 10% threshold is appropriate
- European eel (1.9 % of the precautionary biomass estimate, Section 6.1.6)

For all other species the predicted impingement losses are < 1 %. (For sea lamprey the predicted impingement of <0.13 fish per year is ecologically negligible and could have no effect on the sustainability of the stock. This species has not been further assessed).

7.2 Predicted SZC impingement with LVSE intake heads fitted

With the fitting of LVSE intake heads (designed to reduce the numbers of fish and other organisms being abstracted), the impingement losses of those species that exceeded 1 % in the absence of mitigation were reduced to (Table 12):

- Seabass (0.5 % of SSB)
- Thin-lipped grey mullet (0.9 % of landings) - as an unexploited stock, a 10% threshold is appropriate
- European eel (0.7 % of the precautionary biomass estimate)

The fitting of the LVSE intake heads alone therefore reduces the impingement losses of seabass, thin-lipped grey mullet and European eel to below the 1 % threshold.

7.3 Predicted SZC impingement with FRR systems fitted

With the inclusion of the FRR alone (designed to increase the survival of more robust species), the impingement losses of those species that exceeded 1 % in the absence of mitigation were reduced to (Table 13):

- Seabass (0.7 % of SSB)
- Thin-lipped grey mullet (1.4 % of landings) - as an unexploited stock, a 10% threshold is appropriate
- European eel (0.4 % of the precautionary biomass estimate)

The fitting of the FRR systems alone therefore reduces the impingement losses of seabass and European eel to below the 1 % threshold.

7.4 Predicted SZC impingement with the effect of the LVSE intake heads and FRR systems fitted

With the combined effect of the intake head design and the inclusion of the FRR, the impingement losses of those species that exceeded the 1 % threshold in the absence of mitigation were reduced to (Table 14):

- Seabass (0.28 % of SSB)
- Thin-lipped grey mullet (0.52 % of landings) - as an unexploited stock, a 10% threshold is appropriate
- European eel (0.15 % of the precautionary biomass estimate)

All steps in the mitigation process, including the lower and upper estimates, are given in Appendix C. Given the mitigation of the intake head design and the fitting of the FRR system, no species still exceed the 1 % negligible effects threshold. Further consideration of the predicted SZC effects on conservation species is given in Section 7.6.

7.5 Further consideration of impingement effects on bass and thin-lipped grey mullet

Bass

As described in Section 5 of this report, 20 times more bass were found inshore of the Sizewell-Dunwich Bank in the vicinity of the SZB thermal plume than offshore of the Bank. When SZC begins operation, it will generate a thermal plume but in the deeper water at the SZC outfalls there will be negligible warming at the seabed and the thermal plume effects will be limited to the top 1 m of the sea surface. The SZC plume will have the effect of further warming the inshore waters inshore of the Bank. Bass is a demersal species, but it is known to feed at the surface at night and so could be attracted to the SZC surface plume at night. However, at the surface bass would be invulnerable to the impact of SZC abstraction by the seabed mounted intakes. At depth the water inshore of the Bank would be appreciably warmer than at SZC and there is no reason to consider that the distribution of bass would materially change from what it is now. Making a precautionary assessment that 90% of bass would remain inshore of the Bank (rather than the measure 95%) the expected bass impingement at SZC is 0.028% SSB and not the 0.28% SSB described in Section 7.4.

Thin lipped grey mullet

There is not a directed commercial fishery for grey mullet in the southern North Sea and therefore the landings data (120 t) will substantially underestimate the SSB. The mean length in the commercial catch has been estimated to be in the range 36 to 42cm. At this size the natural mortality is in the range of 0.5 to 0.4 (BEEMS Technical Report TR383) and the calculated sustainable harvesting rate is approximately 33% - 39% SSB (Section 6.1.1). Mullet impingement numbers at SZB show no trend over the period 2009-2017 and provide no evidence that fishing on the stock is unsustainable. It is therefore considered unlikely that mortality on the stock is 33%+ in the southern North Sea and instead a conservative assumption has been made that landings represent 20% of SSB. Using this figure, the predicted impingement in Section 7.4 at 0.5 % of landings is equivalent to approximately 0.1% SSB i.e. below the screening threshold of 1 % of SSB. For such a species without a directed fishery, the use of a 1% threshold is itself overly precautionary and a 10%

threshold is indicated by internationally accepted fisheries assessment methodology (Section 6.1), further reducing the significance of the predicted impingement effect.

The effects of these two changes on the predicted SZC impingement assessment with LVSE intakes and FRR systems fitted is shown in

Table 15.

Table 11 Annual mean SZC impingement predictions with no impingement mitigation. Losses have been converted to adult equivalent (EAV) numbers and weights (t) and calculated as a % of either the mean stock SSB (t) or mean international landings (t). Species where the impingement weight > 1 % of the relevant stock comparator (either SSB or landings – given in bold) are shaded red. Numbers in red font are estimates of the population numbers (e.g. sand goby, smelt, twaite shad, allis shad) or catch numbers (salmon & sea trout)

Species	Mean SZC prediction	EAV number	EAV weight (t)	Mean SSB	% of SSB	Mean landings (t)	% of landings
Sprat	7,125,393	5,352,978	56.23	220,757	**0.03**	151,322	0.04
Herring	2,555,783	1,827,944	344.87	2,198,449	**0.02**	400,244	0.09
Whiting	1,865,492	664,261	189.86	151,881	**0.13**	17,570	1.08
Bass	575,367	128,861	197.26	14,897	**1.32**	3,051	6.47
Sand goby	381,612	381,612	0.73	205,882,353	**0.19**	NA	NA
Sole	250,059	53,233	11.40	43,770	**0.03**	12,800	0.09
Dab	148,921	66,211	2.70	NA	NA	6,135	**0.04**
Anchovy	73,865	71,952	1.49	NA	NA	1,625	**0.09**
Thin-lipped grey mullet	67,684	5,642	2.93	NA	NA	120	**2.45**
Flounder	38,180	17,631	1.44	NA	NA	2,309	**0.06**
Plaice	25,288	8,734	2.15	690,912	**0.00**	80,367	0.00
Smelt	23,863	18,170	0.30	105,733,825	**0.02**	8	3.56
Cod	16,845	6,049	15.74	103,025	**0.02**	34,701	0.05
Thornback ray	10,802	2,082	6.65	NA	NA	1,573	**0.42**
River lamprey	6,720	6,720	0.53	62	**0.86**	1	47.65
Eel	4,516	4,516	1.49	79	**1.89**	14	10.70
Twaite shad	3,601	3,601	1.13	7,519,986	**0.05**	1	84.60
Horse mackerel	4,077	4,077	0.57	NA	NA	20,798	**0.00**
Mackerel	628	628	0.20	3,888,854	**0.00**	1,026,828	0.00
Tope	64	64	0.44	NA	NA	498	**0.09**
Sea trout	10	10	0.02	NA	NA	39,795	**0.02**
Allis shad	5	5	0.00	27,397	**0.018**	0	1.79
Sea lamprey[1]	5	5	0.01	NA	NA	NA	NA
Salmon	0	0	0.00	NA	NA	38,456	**0.00**

Table 12 Annual mean SZC impingement predictions considering the effect of the intake head design. Losses have been converted to adult equivalent (EAV) numbers and weights (t) and calculated as a % of either the mean stock SSB (t) or mean international landings (t). Species where the impingement weight > 1 % of the relevant stock comparator (either SSB or landings – given in bold) are shaded red. Numbers in red font are estimates of the population numbers (e.g. sand goby, smelt, twaite shad, allis shad) or reported catch numbers (salmon & sea trout)

Species	Mean SZC prediction	SZC prediction (adjusted)	EAV number	EAV weight (t)	Mean SSB	% of SSB	Mean landings (t)	% of landings
Sprat	7,125,393	2,729,025	2,050,190	21.53	220,757	**0.01**	151,322	0.01
Herring	2,555,783	978,865	700,103	132.08	2,198,449	**0.01**	400,244	0.03
Whiting	1,865,492	714,484	254,412	72.72	151,881	**0.05**	17,570	0.41
Bass	575,367	220,366	49,354	75.55	14,897	**0.51**	3,051	2.48
Sand goby	381,612	146,157	146,157	0.28	205,882,353	**0.07**	NA	NA
Sole	250,059	95,773	20,388	4.36	43,770	**0.01**	12,800	0.03
Dab	148,921	57,037	25,359	1.03	NA	NA	6,135	**0.02**
Anchovy	73,865	28,290	27,558	0.57	NA	NA	1,625	**0.04**
Thin-lipped grey mullet	67,684	25,923	2,161	1.12	NA	NA	120	**0.94**
Flounder	38,180	14,623	6,753	0.55	NA	NA	2,309	**0.02**
Plaice	25,288	9,685	3,345	0.82	690,912	**0.00**	80,367	0.00
Smelt	23,863	9,139	6,959	0.12	105,733,825	**0.01**	8	1.36
Cod	16,845	6,451	2,317	6.03	103,025	**0.01**	34,701	0.02
Thornback ray	10,802	4,137	797	2.55	NA	NA	1,573	**0.16**
River lamprey	6,720	2,574	2,574	0.20	62	**0.33**	1	18.25
Eel	4,516	1,730	1,730	0.57	79	**0.72**	14	4.10
Twaite shad	3,601	1,379	1,379	0.43	7,519,986	**0.02**	1	32.40
Horse mackerel	4,077	1,561	1,561	0.22	NA	NA	20,798	**0.00**
Mackerel	628	241	241	0.08	3,888,854	**0.00**	1,026,828	0.00
Tope	64	24	24	0.17	NA	NA	498	**0.03**
Sea trout	10	4	4	0.01	NA	NA	39,795	**0.01**
Allis shad	5	2	2	0.00	27,397	**0.01**	0	0.68
Sea lamprey	5	2	2	0.00	NA	NA	NA	NA
Salmon	0	0	0	0.00	NA	NA	38,456	**0.00**

Table 13 Annual mean SZC impingement predictions with FRR systems fitted (no adjustment for the intake head design). Losses have been converted to adult equivalent (EAV) numbers and weights (t) and calculated as a % of either the mean stock SSB (t) or mean international landings (t). Species where the impingement weight > 1 % of the relevant stock comparator (either SSB or landings – given in bold) are shaded red. Numbers in red font are estimates of the population numbers (e.g. sand goby, smelt, twaite shad, allis shad) or reported catch numbers (salmon & sea trout)

Species	SZC prediction	FRR mortality	EAV number	EAV weight (t)	mean SSB	% of SSB	Mean landings (t)	% of landings
Sprat	7,125,393	7,125,393	5,352,978	56.23	220,757	**0.03**	151,322	0.04
Herring	2,555,783	2,555,783	1,827,944	344.87	2,198,449	**0.02**	400,244	0.09
Whiting	1,865,492	1,026,879	365,649	104.51	151,881	**0.07**	17,570	0.59
Bass	575,367	316,778	70,946	108.61	14,897	**0.73**	3,051	3.56
Sand goby	381,612	78,612	78,612	0.15	205,882,353	**0.04**	NA	NA
Sole	250,059	51,512	10,966	2.35	43,770	**0.01**	12,800	0.02
Dab	148,921	80,196	35,656	1.46	NA	NA	6,135	**0.02**
Anchovy	73,865	73,865	71,952	1.49	NA	NA	1,625	**0.09**
Thin-lipped grey mullet	67,684	37,266	3,106	1.62	NA	NA	120	**1.35**
Flounder	38,180	8,816	4,071	0.33	NA	NA	2,309	**0.01**
Plaice	25,288	5,209	1,799	0.44	690,912	**0.00**	80,367	0.00
Smelt	23,863	23,863	18,170	0.30	105,733,825	**0.02**	8	3.56
Cod	16,845	10,142	3,642	9.48	103,025	**0.01**	34,701	0.03
Thornback ray	10,802	2,225	429	1.37	NA	NA	1,573	**0.09**
River lamprey	6,720	1,384	1,384	0.11	62	**0.18**	1	9.82
Eel	4,516	930	930	0.31	79	**0.39**	14	2.20
Twaite shad	3,601	3,601	3,601	1.13	7,519,986	**0.05**	1	84.60
Horse mackerel	4,077	4,077	4,077	0.57	NA	NA	20,798	**0.00**
Mackerel	628	628	628	0.20	3,888,854	**0.00**	1,026,828	0.00
Tope	64	13	13	0.09	NA	NA	498	**0.02**
Sea trout	10	10	10	0.02	NA	NA	39,795	**0.02**
Allis shad	5	5	5	0.00	27,397	**0.02**	0	1.79
Sea lamprey[1]	5	1	1	0.00	NA	NA	NA	NA
Salmon	0	0	0	0.00	NA	NA	38,456	**0.00**

Note 1: Sea lamprey impingement is predicted to be ecologically negligible and would have no effect on the sustainability of the stock (Section 2.1.1)

Table 14 Annual mean SZC impingement predictions considering the effect of the intake head design and with FRR systems fitted. Losses have been converted to adult equivalent (EAV) numbers and weights (t) and calculated as a % of either the mean stock SSB (t) or mean international landings (t). Species where the impingement weight > 1 % of the relevant stock comparator (either SSB or landings – given in bold) are shaded red. Numbers in red font are either estimates of the population numbers (e.g. sand goby, smelt, twaite shad, allis shad) or reported catch numbers (salmon & sea trout)

Species	Mean SZC prediction	SZC prediction after intake head adjustment	FRR mortality	EAV number	EAV weight (t)	mean SSB	% of SSB	Mean landings (t)	% of landings
Sprat	7,125,393	2,729,025	2,729,025	2,050,190	21.53	220,757	**0.01**	151,322	0.01
Herring	2,555,783	978,865	978,865	700,103	132.08	2,198,449	**0.01**	400,244	0.03
Whiting	1,865,492	714,484	393,295	140,044	40.03	151,881	**0.03**	17,570	0.23
Bass	575,367	220,366	121,326	27,172	41.60	14,897	**0.28**	3,051	1.36
Sand goby	381,612	146,157	30,108	30,108	0.06	205,882,353	**0.01**	NA	NA
Sole	250,059	95,773	19,729	4,200	0.90	43,770	**0.00**	12,800	0.01
Dab	148,921	57,037	30,715	13,656	0.56	NA	NA	6,135	**0.01**
Anchovy	73,865	28,290	28,290	27,558	0.57	NA	NA	1,625	**0.04**
Thin-lipped grey mullet	67,684	25,923	14,273	1,190	0.62	NA	NA	120	**0.52**
Flounder	38,180	14,623	3,377	1,559	0.13	NA	NA	2,309	**0.01**
Plaice	25,288	9,685	1,995	689	0.17	690,912	**0.00**	80,367	0.00
Smelt	23,863	9,139	9,139	6,959	0.12	105,733,825	**0.01**	8	1.36
Cod	16,845	6,451	3,884	1,395	3.63	103,025	**0.00**	34,701	0.01
Thornback ray	10,802	4,137	852	164	0.52	NA	NA	1,573	**0.03**
River lamprey	6,720	2,574	530	530	0.04	62	**0.07**	1	3.76
Eel	4,516	1,730	356	356	0.12	79	**0.15**	14	0.84
Twaite shad	3,601	1,379	1,379	1,379	0.43	7,519,986	**0.02**	1	32.40
Horse mackerel	4,077	1,561	1,561	1,561	0.22	NA	NA	20,798	**0.00**
Mackerel	628	241	241	241	0.08	3,888,854	**0.00**	1,026,828	0.00
Tope	64	24	5	5	0.03	NA	NA	498	**0.01**
Sea trout	10	4	4	4	0.01	NA	NA	39,795	**0.01**
Allis shad	5	2	2	2	0.00	27,397	**0.01**	0	0.68
Sea lamprey	5	2	0	0	0.00	NA	NA	NA	NA
Salmon	0	0	0	0	0.00	NA	NA	38,456	**0.00**

Table 15 Annual mean SZC impingement predictions considering the effect of the LVSE intake heads and FRR systems fitted and the corrections to thin lipped grey mullet and bass assessment detailed in Section 7.5. Losses have been converted to adult equivalent (EAV) numbers and weights (t) and calculated as a % of either the mean stock SSB (t) or mean international landings (t). Numbers in red font are either estimates of the population numbers (e.g. sand goby, smelt, twaite shad, allis shad) or reported catch numbers (salmon & sea trout)

Species	Mean SZC prediction	SZC prediction after intake head adjustment	FRR mortality	EAV number	EAV weight (t)	mean SSB	% of SSB	Mean landings (t)	% of landings
Sprat	7,125,393	2,729,025	2,729,025	2,050,190	21.53	220,757	**0.01**	151,322	0.01
Herring	2,555,783	978,865	978,865	700,103	132.08	2,198,449	**0.01**	400,244	0.03
Whiting	1,865,492	714,484	393,295	140,044	40.03	151,881	**0.03**	17,570	0.23
Bass	57,537	22,037	12,133	2,717	4.16	14,897	**0.03**	3,051	0.14
Sand goby	381,612	146,157	30,108	30,108	0.06	205,882,353	**0.01**	NA	NA
Sole	250,059	95,773	19,729	4,200	0.90	43,770	**0.00**	12,800	0.01
Dab	148,921	57,037	30,715	13,656	0.56	NA	NA	6,135	**0.01**
Anchovy	73,865	28,290	28,290	27,558	0.57	NA	NA	1,625	**0.04**
Thin-lipped grey mullet	67,684	25,923	14,273	1,190	0.62	600	**0.10**	120	0.52
Flounder	38,180	14,623	3,377	1,559	0.13	NA	NA	2,309	**0.01**
Plaice	25,288	9,685	1,995	689	0.17	690,912	**0.00**	80,367	0.00
Smelt	23,863	9,139	9,139	6,959	0.12	105,733,825	**0.01**	8	1.36
Cod	16,845	6,451	3,884	1,395	3.63	103,025	**0.00**	34,701	0.01
Thornback ray	10,802	4,137	852	164	0.52	NA	NA	1,573	**0.03**
River lamprey	6,720	2,574	530	530	0.04	62	**0.07**	1	3.76
Eel	4,516	1,730	356	356	0.12	79	**0.15**	14	0.84
Twaite shad	3,601	1,379	1,379	1,379	0.43	7,519,986	**0.02**	1	32.40
Horse mackerel	4,077	1,561	1,561	1,561	0.22	NA	NA	20,798	**0.00**
Mackerel	628	241	241	241	0.08	3,888,854	**0.00**	1,026,828	0.00
Tope	64	24	5	5	0.03	NA	NA	498	**0.01**
Sea trout	10	4	4	4	0.01	NA	NA	39,795	**0.01**
Allis shad	5	2	2	2	0.00	27,397	**0.01**	0	0.68
Sea lamprey	5	2	0	0	0.00	NA	NA	NA	NA
Salmon	0	0	0	0	0.00	NA	NA	38,456	**0.00**

7.6 Consideration of the impingement losses of finfish species of conservation concern

7.6.1 Cucumber Smelt

Smelt (*Osmerus eperlanus*) are found in coastal waters and estuaries around the western coast of Europe, from southern Norway to north-west Spain (Maitland, 2003a). Although there are several non-migratory populations in large freshwater lakes in Scandinavia, it is usually found in coastal waters and migrates into large clean rivers to spawn (Wheeler, 1969). Adults live in the marine environment, but migrate to estuarine or slightly brackish rivers in early spring (February to April) to spawn, after which the adults return to sea (Maitland, 2003a). Smelt shed their adhesive eggs onto the river bed in the brackish reaches of tidal rivers during March and April, where they hatch in about 3−4 weeks. Spawning appears to be determined by temperature and tides. In the River Thames, spawning takes place in the Wandsworth area of the estuary and 0+ fish first appear at 18mm at Greenwich in mid-May (Colclough et al., 2002).

The smelt was once common in Great Britain and supported commercial fisheries in the estuaries of most large rivers from the Clyde and Tay south. Maitland (2003a) reports that fisheries for smelt existed in the tidal reaches of all the Broads rivers in Norfolk until at least 2002; commercial fisheries ‘yielding 3 to 6 t’ per annum were still active in the River Waveney in 1991; smelt are occasionally taken in herring nets in the Orwell Estuary; and commercial fishermen were taking large catches – 190−250 kg per day in the Medway and the River Thames by 2002. Today, smelt occur in at least 36 water courses in England and Wales, with large populations in the rivers Thames, Humber and Dee, the Wash and Great Ouse, as well as in water courses of the Norfolk Broads. Smaller populations exist in the rivers Alde/Ore, Ribble and Conwy, and recovery of supposed extinct populations seems to be underway in the rivers Tyne and Mersey (Colclough and Coates, 2013).

There are commercial fisheries for smelt in the Rivers Waveney, Bure and Yare, predominantly for angling baits, although smelt are now sold to restaurants (Dr. A. Moore, Cefas, pers. comm.) and since 2011 there has been a requirement for commercial smelt fisheries to be authorised by the EA and to make annual catch returns. The annual catch of smelt in 2014 was 11,006 kg from 4 licence holders (EA, 2015). However, the report does not state the rivers that the licence holders exploited, but it is known that they are based in the Ouse (Yorkshire and Cambridge), Waveney/Yare and Thames (Dr. A. Moore, Cefas, pers. comm.).

Smelt are found all along the Anglian coast, in the southern North Sea and on the European coast from the Channel to Denmark but there is no targeted fishery at sea. The nearest estuary to Sizewell with a known smelt population is the Alde/Ore, approximately 25 km to the south of Sizewell. Other than that, the nearest estuary to Sizewell is the Blyth at approximately 12 km to the north of Sizewell. Adult smelt have been sampled in the Blyth but there is no evidence of a breeding population. Surveying in April and May 2016 found no evidence of suitable spawning habitat, a barrier to upstream migration, no eggs nor any smelt in spawning condition at the time that other Anglian rivers contained spawning aggregations (BEEMS Technical Report TR382). This work concluded that it was highly unlikely that there was a spawning population in the Blyth primarily due to a lack of suitable spawning habitat and the presence of a barrier to up river migration.

Information on smelt stocks is limited. Colclough and Coates (2013) concluded that the smelt found in the Wash are probably from a common stock which may access some or all of the tributaries that flow into the Wash, and Maitland (2003a) reported that it is likely that stocks in Suffolk belonged to a population associated with the Norfolk Broads and the estuarine and brackish waters around Great Yarmouth and Lowestoft. More recent genetic analysis of 215 smelt collected from the SZB CIMP programme and from the Thames, Waveney, Great Ouse and Tamar estuaries showed that East Anglian smelt are genetically homogeneous with no genetic structuring seen within the region. Smelt from the Tamar was clearly distinct from the East Anglian collections (BEEMS Technical Report TR423).

Given the genetic information on the smelt at Sizewell, it is probable that the smelt impinged are from multiple locations on the east coast of the UK and, based on the comparable distances, from European estuaries of at least the Scheldt (Belgium) and the Elbe in Germany (this hypothesis is considered reasonable but it is recognised that it has not yet been proven). Considering only UK populations and given the limited number of licences issued for commercial exploitation, the size of fishery landings will be a substantial underestimate of the stock size. Comparisons have therefore been made against estimates of population size for the River Elbe (Data on the abundant Scheldt population could not reliably be extracted form the available publications and would require clarifications from the Belgian authorities before their data could be used in the SZC assessment). Between 2009 and 2017, an estimated annual average 105.7 million adult smelt passed through the River Elbe (BEEMS Scientific Position Paper SPP100). In the absence of mitigation, the losses of smelt by SZC represent 0.02 % of these population numbers; with consideration of the LVSE intake head design losses fall to 0.01 % of the population.

Smelt abundance at Sizewell as indicated by the SZB impingement data has no trend from 2009 to 2017 and has apparently not changed since 1981/82 (Section 9.1). Losses due to commercial fishing in the 35+ year period has not had any discernible effect on smelt numbers at Sizewell and fishing mortality must therefore have been low with SSB being much greater than UK landings. If the possibility of smelt at Sizewell being part of a southern North Sea population embracing Belgian and German sub populations is ignored, assessment of SZC entrapment effects would have to be based upon UK landings data. On a conservative basis the maximum sustainable harvesting rate on an 'Anglian' smelt stock would be 16% (using the precautionary assumptions in Section 6.1.1 for calculating the sustainable harvesting rate for short lived species (i.e. the use of natural mortality/4) and an assumed natural mortality of 1.0 for 100% mature 2-year old fish, BEEMS Technical Report TR383). Given the stability of the Sizewell population as indicated by impingement numbers, fishing and other anthropogenic mortality is most likely less than 16% SSB and SSB will therefore be 6.25 times UK landings, possibly more given the very restrictive licensing policy operating in the UK (e.g. with only 4 licences issued in 2015). I,e. the indicated effect level of SZC entrapment in Table 18 of 1.36% of landings would equate to 0.22% SSB and be assessed as negligible.

7.6.2 River lamprey and sea lamprey

The river lamprey (*Lampetra fluviatilis*) is found in coastal waters, estuaries and accessible rivers in western Europe, from southern Norway to the western Mediterranean. It is widespread in catchments throughout the UK, except in northwest Scotland and in industrial areas where water quality is poor or where obstacles prevent the upstream migration of adults prior to spawning (Maitland, 1972). The rivers of the Severn Estuary are thought to be the most important area in the UK for sea lamprey and possibly river lamprey too (Bird, 2008).

The biology and ecology of lampreys have been described in detail and reviewed by Maitland (2003b). Both sea and river lampreys spawn in coarse, well aerated river beds and juveniles, known as ammocoetes, spend several years living in aerobic silt beds filtering sediments, before transforming to migrants that move downstream to sea in the spring. After some years growing in the marine environment they move back into freshwater, migrate upstream to spawn. River lamprey move into freshwater during the previous summer, winter and spring before spawning in spring, whereas sea lamprey migrate into estuaries in the spring and then upriver to spawn in late spring to early summer (Hardisty, 1986). The fish are semelparous and die after spawning (Larsen, 1980).

Genetic studies suggest that sea lamprey (*Petromyzon marinus*) are a single, pan-European population (Almada et al., 2008) with widespread distribution. This is thought to be determined to a large extent by the movements of the fish hosts on which the lampreys feed and the fact that the adult lamprey do not display any apparent homing behaviour during spawning migrations (Berstedt and Seelye, 1995). River lamprey also display the same parasitic behaviour in the marine environment and are also not considered to home to natal rivers. As would be expected they show only low levels of genetic differentiation between local stocks across England. (Bracken et al., 2015).

Fisheries for river lamprey exist in the Netherlands, Sweden, Finland, Estonia and in the UK. Sea and river lampreys are qualifying features of the Humber SAC. River lamprey are a primary reason for selecting the Derwent SAC, with sea lamprey a qualifying feature.

Despite a historical long-term decline in status, the distribution of river lamprey in waters of England and Wales appears to have increased in recent years, coincident with increased water quality in UK and European rivers. However, recording effort has also increased and JNCC, 2013 considered that it remained unclear whether the apparent increase in range is a consequence of increased levels of reporting or a true change in status. Results from impingement measurements at Sizewell showed no river lamprey caught in the years1981/82 at SZA but an estimated annual mean of 2624 caught at SZB (Section 9.1). At Sizewell there has, therefore, been a substantial increase in river lamprey population sometime in the period 1982 to 2009. This is mirrored in multinational North Sea ecoregion monitoring surveys which found an almost complete absence of lampreys from 1977 to 1992 followed by a substantial increase in numbers thereafter. (Heesen *et al* 2015).

Historically, commercial fisheries for river lamprey have taken place in several large rivers, including the Severn and Yorkshire Ouse. The Ouse bait fishery was re-established in 1995, with annual catches around 4000 kg (Masters et al., 2006) which was estimated to have amounted to approximately 20% of the Ouse population but without showing any evidence of population decline. Since 2013, the Environment Agency has controlled the fishery by means of authorisations: 4 for 2015 and 2016 (EA, 2017a, 2017b). This fishery is subject to a catch limit (JNCC, 2013), and an overall agreed anthropogenic loss limit for river lampreys of 5 % of the population size estimated from mark-recapture experiments carried out in 2003 (Masters et al., 2006). There is also a limit on the season (open: 1 November to 10 December). Total allowable catch (TAC) has varied over the years and is often different for various rivers of the Humber catchment (Dr J Masters, Environment Agency, pers. Comm.). In 2017 and 2018 fishing was allowed on a non-consumptive basis only (catch and release), but for 2019 a quota of 898 kg has been set for the Humber catchment.

In 2018, the Hull International Fisheries Institute (HIFI) undertook mark-recapture experiments of lampreys in the commercial fishery at Naburn, the River Ouse. Almost 1,500 lampreys were marked with PIT tags were used to estimate the run size of the lamprey population in the Humber catchment. Work was carried out during the fishing season only, but lampreys can migrate between October and February, so the estimated run size is likely an underestimation of the total spawning run. The Environment Agency used the HIFI data to create a population estimate for the Humber catchment of 783,043 individuals, equating to 61.86 t.

SZC with the proposed intake head design and FRR mitigation is predicted to take 530 individuals or 0.04 t of river lampreys, which equates to 0.07 % of the estimated 2018 lamprey run in the Humber catchment. The Southern North Sea population of river lamprey are probably one stock with spawning taking place in the Ouse in the UK but also in the Scheldt in the Netherlands where the adult population is estimated to be in the 100,000s (Jansen et al., 2007) and in other European rivers that drain into the North Sea (e.g. rivers Eider, Elbe, Weser and Ems). The SZC predicted impact on the river lamprey population is, therefore, considered negligible.

For sea lampreys, the estimated annual impingement loss with the FRR mitigation is < 0.13 fish, with a maximum of 2 individuals. This is considered negligible for a stock that is widespread throughout the N. Sea.

7.6.3 Twaite shad and allis shad

Allis shad (*Alosa alosa*) and twaite shad (*Alosa fallax*) both belong to the herring family and historically had a broad distribution along the Northeast Atlantic coast. Both species are anadromous; adults spend most of their lives in the marine environment but migrate through estuaries to spawn in freshwater. Populations of both species have declined, their distribution has diminished, and they are both classified as species of conservation concern. Both are listed in Appendix III of the Bern Convention and Annexes II and V of the Habitats Directive.

Alosa alosa was historically distributed along the eastern Atlantic seaboard from Norway to North Africa and in the western Mediterranean. It has declined significantly throughout its range and is now extinct in several former areas. Currently, populations of *A. alosa* exist along the north-eastern Atlantic coasts in some large rivers of France (Loire, Gironde–Garonne–Dordogne, and Adour) and Portugal (Minho and Lima) (BEEMS Scientific Position Paper SPP071/s). There are currently no known spawning sites for this species in the

United Kingdom, and only two locations in the UK where individuals in breeding condition have been recorded: the river Tamar in SW England and the Solway Firth on the border between England and Scotland (Jolly *et al*., 2012). Immature adults are occasionally found in the Bristol Channel, the English Channel and the UK east coast. It is considered probable that British-caught specimens are from the Loire to Gironde populations (BEEMS Scientific Position Paper SPP071/s). *A. alosa* only spawns in any substantial numbers in France and Portugal (the species has recently been reintroduced into the Rhine but the number of recruits is still small). There is no international stock assessment for *A. alosa* but some assessments are performed on specific French watersheds. The Gironde–Garonne–Dordogne basin had a notable commercial fishery at the end of the 20th century. The adult population (age 4+) was estimated to be 710 000, 798 000, and 834 000 in 1994, 1995, and 1996, respectively, with a mean exploitation rate by the commercial fishery of 44%. It was reported that the commercial fishery in that basin caught approximately 500 t annually. However, in the first decade of the 21st century, there was a recruitment collapse probably due to over fishing and a fishery moratorium was imposed in the Gironde estuary from 2008 (BEEMS Scientific Position Paper SPP071/s). The estimated adult stock size in the basin was 27,397 in 2009. The Loire watershed also has a breeding population of *A. alosa* and a small commercial fishery. The count of alosa was 2,557 in 2009 but the video counting system does not cover all the tributaries of the Loire and cannot distinguish between *A. alosa* and *A. fallax*. The counters are located relatively high in the river basin at ranges of 260 – 663km from the sea and are, therefore, probably counting mostly *A. alosa*. It is also known that a substantial amount of spawning takes place downstream of the counters thereby underestimating adult numbers (BEEMS Scientific Position Paper SPP071/s).

A. fallax is distributed along most of the west coast of Europe from the eastern Mediterranean Sea to southern Norway and in the lower reaches of large rivers along these coasts that are accessible to the fish (i.e. rivers that lack barriers to migration). The species has declined substantially across Europe and in the UK; it is now known to breed only in the Severn River Basin District (RBD – in the Severn, the Wye, the Usk and the Tywi) and in the Solway Firth. There are also non-breeding populations in the UK off the southern and eastern coasts, at Looe Bay, Hastings and Sizewell (Jolly et al., 2012). The decline of the *A. fallax* population has not been as severe as that of *A. alosa*, probably because of its ability to use spawning sites closer to the sea than those of *A. alosa*; sites that are not, therefore, subject to the barriers to migration that block *A. alosa* from accessing its traditional spawning sites.

Allis shad are protected under the Wildlife and Countryside Act (1981), and it is illegal to fish for the species in the UK.

Most of the twaite shad reported are taken as bycatch in coastal or estuarine net fisheries or by anglers fishing for other species (although several hundred are usually taken each year by anglers in the River Wye and other rivers into which twaite shad run to spawn).

Shad of both species migrate into rivers and spawn in flowing water over stones and gravel from mid-May to mid-July. Allis shad normally spawn only once, but twaite shad may spawn several times in their lives (Miran Aprahamian, Environment Agency, unpublished).

Young twaite shad remain in estuaries to feed on invertebrates, initially insect larvae and zooplankton and then increasingly on larger crustaceans such as shrimps and mysids and also small fish as they grow, reaching 10–15 cm after one year (Aprahamian, 1989). Water temperature during the months June−August seems to be an important influence in determining year-class strength in 0+ twaite shad.

Little is known about the movements of adult shad along the coast, but they have been studied when they enter estuaries and migrate upstream to spawn in freshwater, in particular in relation to barriers caused by obstacles such as waterfalls or man-made dams and weirs, and pollution (Aprahamian, 1982; Maitland, 1972). Adult shad of both species gather in the estuaries of suitable rivers in April and May, and the upstream migration from the estuary appears to be modified by temperature, with peak migratory activity at 10–14ºC and during relatively high discharge levels (Aprahamian, 1982; Claridge and Gardner, 1978).

On the North Sea coast of Europe adult twaite shad are relatively common from the Belgium to Denmark. Adult populations are increasing in the known German spawning rivers of the Elbe/Weser with sporadic spawning in the Ems (Helcom, 2013; Magath and Thiel, 2013). Twaite shad populations are also increasing in the Baltic, particularly in Poland and Lithuania where the species is classified as in good condition with increasing populations (Helcom, 2013). Genetic analyses of twaite shad from Sizewell demonstrate that they do not originate from the Severn catchment (Jolly et al., 2012). Sabatino and Alexandrino (2012) identified a North Sea twaite shad population with low genetic diversity between fish sampled off Belgium (Scheldt) and Denmark and also the Solway Firth. These analyses identified separation between the Baltic and North Sea populations and the North Sea population would therefore appear to most likely originate from the German rivers of the Elbe/Weser and the Belgian river Scheldt. The twaite shad caught at Sizewell are >1 yr old juveniles to sexually mature adults that are part of widely dispersed feeding population in the North Sea before eventually returning to probably European rivers to reproduce.

Figure 5 shows a map of Natura 2000 sites designated for *A. fallax* in the southern North Sea and eastern Channel (https://eunis.eea.europa.eu/species/Alosa%20fallax downloaded November 2019). There are no known UK east coast spawning sites nor HRA designated sites for the species. European sites designated for the species include the estuarine and coastal areas in which the species either feeds as juveniles or transits on its way to southern North Sea feeding grounds as adults. (For example, the entire German North Sea coast is in one of more designated sites) More materially the designated sites include the European rivers where the species is known from monitoring data to successfully spawn. The Elbe in Germany has largest breeding population with other breeding populations in the Scheldt (Belgium) and sporadically in the Weser (Germany). The other rivers shown in Figure 5 have negligible or no spawning currently.

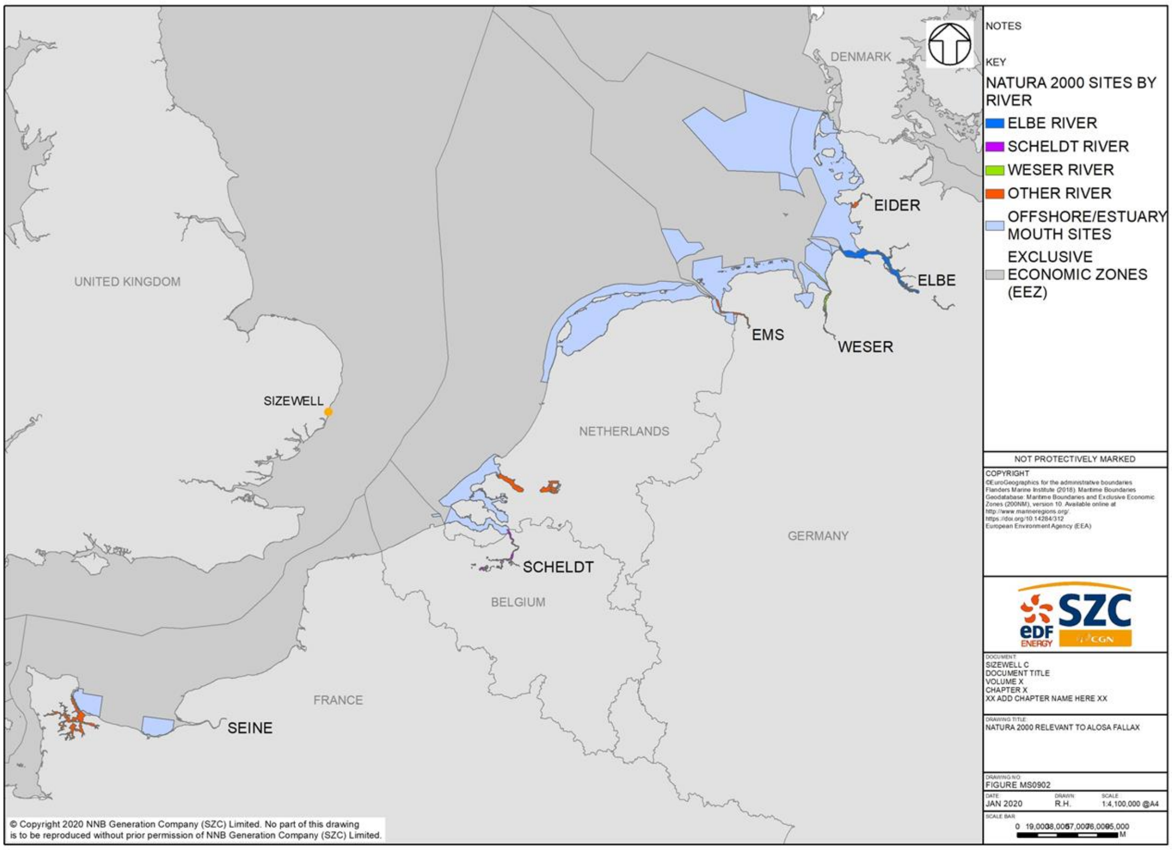


Figure 5 Southern North Sea and Channel Natura 2000 sites designated for Twaite shad

An estimated 3,601 twaite shad will be lost by the proposed SZC station in the absence of any mitigation. This number is reduced to 1,379 individuals with the effect of the LVSE intake heads (FRR mitigation does not increase shad survival as the group is classed as pelagic). There is currently no SSB for the North Sea twaite shad population and no directed fishery, so comparison with landings data does not provide a meaningful assessment. Losses have been compared with the population estimates available from spring monitoring surveys conducted on the Rivers Elbe and Scheldt. Between 2009 and 2017, an average estimated 7.5 million adult twaite shad pass through these two river systems (BEEMS Scientific Position Paper SPP100). In the absence of mitigation, the losses of twaite shad by SZC represent 0.05 % of these population numbers; with consideration of the LVSE intake head design losses fall to 0.02 % of the population. This represents a negligible effect on the species in these 2 river systems which would have no effect on the sustainability of the population. It is therefore considered that there would be no significant transboundary effects on any European site designated for *A. fallax*.

7.6.3.1 Identification of 0 group twaite shad in SZB impingement samples

0 group twaite shad have some similarities with juvenile herring (e.g. the scale pattern is similar) which can make identification difficult and the accuracy of identifying 0 group twaite shad has been queried. For the impingement sampling at SZB a very experienced team of fish taxonomists has been deployed including those who have handled juvenile twaite shad from Hinkley Point (where 0 group fish do occur). Based upon published taxonomic information and the experience with the Hinkley Point samples, we are confident that the sampling team can reliably distinguish twaite shad from other clupeids (based upon a combination of markings, body depth and eye size).

7.6.4 European eel

The European eel (*Anguilla anguilla*) is distributed across the majority of coastal countries in Europe and North Africa, with its southern limit in Mauritania (30°N) and its northern limit situated in the Barents Sea (72°N) and spanning the entire Mediterranean basin (WGEEL 2019).

The European eel life history is complex, being a long-lived semelparous (mature adults die after spawning) and widely dispersed stock. The shared single stock is genetically panmictic and data indicate the spawning area is in the southwestern part of the Sargasso Sea. The newly hatched leptocephalus larvae drift for two to three years with the ocean currents for more than 5000km to the continental shelf of Europe and North Africa, and enter continental waters. There, they metamorphose into the post-larval transparent glass eel. At this stage, glass eels migrate across the continental shelf to the coast. After reaching the coast, glass eels enter estuaries (in the UK Severn starting from about January/February). Glass eels metamorphose into pigmented elvers which either remain and feed in coastal marine or estuarine waters or begin active upstream migration to freshwater. The growth stage, known as yellow eel, may take place in marine, brackish (transitional), or freshwaters. This stage may last typically from two to 25 years (and could exceed 50 years) prior to metamorphosis to the “silver eel” stage and maturation. At this stage silver eels migrate 5000 km+ back to their spawning grounds. Age-at-maturity varies according to temperature (latitude and longitude), ecosystem characteristics, and density-dependent processes. The European eel life cycle is shorter for populations in the southern part of their range compared to the north. Age at maturity ranges from 10 to 20+ years in northern temperate waters (Vøllested, 1992), and is earlier for males than for females. (WGEEL 2019, McCleave 1993, Tesch 2003, Harrison *et al* 2014).

The stock of the European eel is described as being outside safe biological limits, with urgent action required by European Union Member States to assist recovery of the panmictic stock (Harrison *et al* 2014). Historical publications indicate that the decline in stock abundance and/or fishing yield might have started as early as in the 1800s, and might have been related to inadvertent side effects of anthropogenic actions (water management). The downward trend in yield has been acknowledged internationally since the late 1960s, but up to today, it is unclear what processes were causing the decline, which occurred even in times of high recruitment up to 1980 (Dekker and Beaulaton 2015).

7.6.4.1 Eel catch statistics (from WGEEL 2019)

Glass eels

Glass eel fisheries within the EU take place in France, UK, Spain, Portugal and Italy. Glass eel landings have declined sharply from 1980, when landings were larger than 2000 tonnes, to 62.2 t in 2018, 58.6 t in 2019 (provisional figure), and a mean for the previous 5 years (2013–2017) of 56.5 t. The amount of glass eel arriving in continental waters declined dramatically in the early 1980s to a low point in 2011. The reasons for this decline are uncertain but may include commercial overexploitation, pollution, non-native parasites, diseases, migratory barriers and other habitat loss, mortality during passage through turbines or pumps, and/or oceanic factors affecting migrations. Statistical analyses of time-series from 1980–2019 show that there was a change in the trend of glass eel recruitment indices in 2011; the recruitment has stopped decreasing and has been increasing in the period 2011–2019 with a rate statistically significantly different from zero. The highest point during the period from 2011–2019 was in 2014. It is not yet clear whether this change indicates a sustained recovery in eel recruitment.

Yellow and silver eels

Total EU commercial landings of yellow and silver eels were estimated to be around 20,000 t in the 1950s to 2000–3500 t around 2009, most recently being 2393 t in 2017, 2694 t in 2018 (provisional) and a mean of 2729 t for the preceding 5 years (2012–2016).

Recreational catches and landings are poorly reported, and so values must be regarded as minima. Recreational landings for yellow and silver eel combined declined from 543 t for 2016 (from ten EU countries), 195 t for 2017 (eight countries reporting) and 148 t for 2018 (five countries reporting). Overall, the impact of recreational fisheries on the eel stock remains largely unquantified although landings are considered to be at a similar order of magnitude to those of commercial fisheries.

A rough estimate of eel loss to all non-fishery anthropogenic factors (largely hydropower and pumps in rivers) estimated from reported mortality indicators from approximately half of the countries that report eel statistics to the EU is 1625 tonnes annually.

Summarising these statistics, the annual EU anthropogenic impact on the yellow and silver eel stock in 2016 was approximately 4,900 t (comprising 2729 t fishing, 543 t recreational fishing and 1625 t from pumps and hydropower in rivers) compared with approximately more than 22,000 tonnes in the 1950s when commercial fishing accounted for 20,000 tonnes

There is no internationally agreed SSB estimate for the entire eel stock.

7.6.4.2 Local eel population

There was once a considerable fishery for yellow and silver eels in the Anglian River Basin District (RBD), the catchments of rivers that drain to the North Sea along the east coast of England between the Humber and Thames (Defra, 2010). However, eel fishing is now mainly a subsistence activity for the remaining fishers in Suffolk rivers and the adjacent coast, who use fyke nets for yellow eels between March and November and for silver eels from September to December.

Although yellow eel populations in freshwater catchments may be considered as largely separate, there is no evidence that silver eels migrating outwards or inward-recruiting glass eels comprise separate biological entities, i.e. the European eel is a single stock unit (Maes et al., 2006; Palm et al., 2009). ICES (ICES, 2013) demonstrated that the overall recruitment of European glass eels is now <10% of the levels observed in the 1960s and 1970s, but there is no assessment of the whole stock. In 2008 and again in 2013, the European eel was listed in the IUCN Red List as a critically endangered species (Jacoby and Gollock, 2014).

The most comprehensive assessment available for the status of the eel population in East Anglia is provided in the Eel Management Plan (EMP) for the Anglian River Basin District (RBD). Data on yellow eel populations for Essex and Suffolk catchments are derived from electric fishing surveys, carried out as part of the EA's routine monitoring programme. An assessment, based on combined data gathered from 2009 to 2011 and reported to the European Commission in June 2012 as part of the UK's EMP Progress Report, estimated a total output of 62.3 t of silver eels from the Anglian RDB each year (Defra, 2012). Estimates were updated and presented for individual years in 2015 and 2018 (Defra, 2018, 2015).

7.6.4.3 Potential risks of SZC eel entrapment

In principle, the following eel life stages could be at risk from entrapment in SZC:

- Glass eels migrating along the coast from the north or the south to find a suitable estuarine/freshwater habitat;
- yellow eels moving between different river systems along the East Anglian coast or living in coastal waters; and
- silver eels migrating to the Sargasso Sea that after leaving estuaries along the East Anglian coast may transit past Sizewell.

Glass eels

Glass eels enter estuaries all year round, with migration peaks depending on latitude and also the variability of oceanic factors. In southwest Spain, highest densities occur between late autumn and spring with two migration peaks observed, whereas peak glass eel migration in the UK is later, typically occurring from February to May.

Glass eels that contribute to UK populations first arrive in the Western Approaches and then transit with the tidal currents either through the English Channel into the southern North Sea or from the north, following currents that flow around Scotland and southwards into the southern North Sea. The time to reach the southern North Sea is dependent on met-ocean conditions over Northern Europe and the relative strength of the Gulf Stream and associated currents around the British Isles. However, little is known about the residence times of glass eels in the southern North Sea. It is considered that glass eels reach the coast and then seek a salinity or other chemical cue to commence migrations up estuaries and then, for a large proportion of their number, to freshwater. The time spent in the open North Sea will, therefore, be dependent on the tidal currents and when the eels sense estuarine cues. In the journey from the Western Approaches to the southern North Sea the density of glass eels in coastal waters will be reduced progressively and substantially as large proportions of the eels migrate up estuaries encountered on route. In particular, eels travelling through the Channel and then heading north will encounter the very large Thames freshwater signal followed by signals from Essex and Suffolk rivers before they reach the coast in the vicinity of Sizewell. Residual hydrodynamic flows will also tend to carry a proportion of eels passing through the Straits of Dover towards the Dutch Coast. Eels migrating from the north will also encounter freshwater signals at for example the Humber, the Wash, North Norfolk coast rivers and the Broads at Yarmouth. Thereafter residual flows will tend to carry eels towards continental Europe. The net result of these tidal flow patterns is that the expected glass eel density in the vicinity of Sizewell would be expected to be amongst the lowest in the UK.

Given their morphology of typically 4 mm width (and up to 8mm for 130 mm elvers), it is likely that most glass eels will pass through the 10 mm mesh on the SZB and proposed SZC cooling water screens and only rarely appear in impingement samples. In the BEEMS CIMP programme from 2009 to 2017 two glass eels have been sampled; 1 in March 2013 and 1 in January 2017 with both of length of approximately 67.5mm. The BEEMS targeted glass eel surveys in April and May 2015 only detected 1 glass eel in 105 valid tows using a methodology which successfully sampled many glass eels in the Bristol Channel. No glass eels or elvers have been detected in water drawn from the SZB forebay during the 12 month BEEMS Comprehensive Entrainment Monitoring Programme at Sizewell in 2011 (BEEMS Technical Report TR318) nor in any of the very large number of plankton surveys conducted at Sizewell. The totality of data from this extensive sampling programme led to the conclusion in BEEMS Technical Report TR318 that whilst glass eels are present in Sizewell coastal waters, that their density was very low at this location. The potential impact of glass eel entrainment in SZC was therefore assessed as negligible, especially given the high measured survival in laboratory studies that mimic the physical and chemical conditions and time of exposure and that any entrained glass eels would encountered within the power station (BEEMS Technical Report TR318).

During stakeholder engagement on the effects of SZC on fish, stakeholders have questioned whether the BEEMS surveys would have adequately detected glass eels. In particular, it has been suggested that:

1. the glass eel specific surveys targeted the surface waters during daylight when glass eels would have been seeking refuge near the seabed.
2. all of any glass eels sampled in the entrainment monitoring programme could have crawled out of the sampling nets before the nets were emptied.

These points have been carefully considered. In relation to point 1, the eel behaviour described is from upper estuaries not in lower estuaries or coastal waters (Harrison et al., 2014). At Hinkley Point (i.e. in the lower estuary) large numbers of glass eels were successfully sampled using the same methodology employed at Sizewell of sampling in the upper part of the water column during the daytime on the flood tide. As expected virtually no glass eels were detected on the ebb tide which is the behaviour that would be expected from fish using selective tidal stream transport (STST) to migrate on all of the available flood tides. This confirms that glass eels do migrate during daylight hours on flood tides at such locations and not just at night. This is supported by Lambert et al., (2007) where 30% of glass eels were found to migrate on the flood by day in the lower section of the Gironde estuary. Glass eels have poor swimming abilities and STST is the most energy efficient means of transport. Based upon the relative timings of eel arrivals in UK estuaries it is considered likely that glass eels employ STST on flood tides in coastal waters during day and night, particularly where underwater light levels are low due to high suspended sediment concentrations such as on the UK east coast. The question of whether the BEEMS surveys were too late could also be posed. The sample timings were determined from known glass eel arrival times on the UK east coast. For example, Environment Agency monitoring at Beeleigh Cut on the Blackwater (i.e. to the south of Sizewell where glass eel densities would be expected to be higher than off Sizewell assuming their likely southern migration route around the UK) showed that peak glass eels numbers were detected in May with substantial numbers in April and June but very low numbers of arrivals in March which would imply that the BEEMS survey timing (in April and May) was appropriate. The year 2015 could also have been a year of anomalously low glass eel recruitment but Figure 6 shows that this was not the case and in fact 2015 was a year that reflected the increase in glass eel recruitment observed across Europe from 2011 onwards described in Section 7.6.4.1. The BEEMS targeted glass eel surveys are, therefore, not considered invalid and if substantial number of glass eels were present at Sizewell the surveys would have detected them.

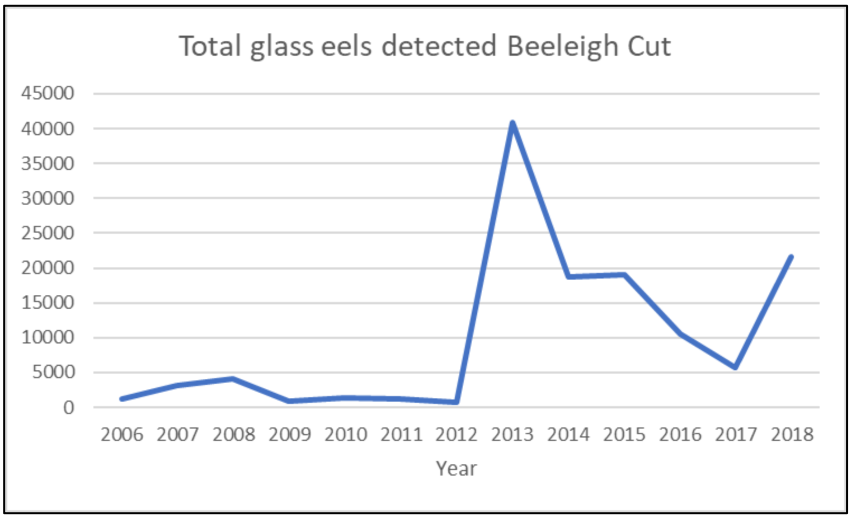


Figure 6 EA monitoring data for upriver glass eel migration from the Blackwater in Essex.

In relation to the second point, glass eels are indeed known to crawl to overcome barriers but this is reported behaviour from upper estuaries:

"*Although STST is the primary mechanism facilitating migratory passage through estuaries, where tidal effects become weaker in upper estuarine zones, a behavioural shift to active swimming is necessitated to effect further dispersion upstream at the freshwater interface or more certainly from the point where they accumulate glass eels change their behavioural pattern and actively migrate counter current. Such an active migration is revealed in the 'crawling' behaviour that glass eels display on trapping ladders."* Harrison et al., 2014.

The crawling behaviour described by stakeholders is a behavioural change associated with the tidal interface not in the coastal zone. Even if glass eels in the marine environment can climb barriers, the net sides in the entrainment sampling tanks were very steep and Cefas considers it highly unlikely that, if any eels had been caught, that all would have climbed out of all of the many nets deployed. In particular, the pump used to sample the SZB forebay for the entrainment sampling was not selected to ensure the survival of glass eels and the small impellor size would have delivered glass eels either moribund or dead into the sampling nets, substantially reducing the possibility of escapement from the sampling nets. In addition, the sampling pump was inspected after each of the 40 entrainment sampling events and no eels were found within the pump or its strainer. On a weight of evidence based approach, we are therefore confident that any glass eels caught in the entrainment sampling system would have most likely been retained by the sampling system and not have escaped.

In conclusion, the glass eel migration pattern around the UK, the strength and direction of coastal currents and the large number of freshwater rivers that the eels would encounter on route would mean that glass eel densities at Sizewell would be expected to be very low and amongst the lowest on the UK coast (on the eel migration route). This low density conclusion is supported by monitoring data. However, that monitoring data does confirm that a few glass eels do transit past Sizewell whilst seeking freshwater signals. On energy efficiency grounds this migration is most likely to use a form of STST in near surface waters. When the tide is in the 'wrong' direction the evidence suggests that glass eels are stationary on, or even buried in the bottom sediments to avoid being carried away from their preferred migration course. Such a migration strategy will mean that there is a low risk of abstraction into power stations with bottom mounted intakes which do not abstract surface water except minimally at slack water. The deeper the intakes, the lower the risk of abstraction. It would therefore be expected that glass eel abstraction at SZB would be greater than at SZC due to substantially deeper water at the proposed SZC intakes. The abstraction risk zone for the SZC intake heads depends on the swimming ability of the species. Glass eels are weak swimmers and can sustain approximately 0.25 m/s for only 3 minutes before exhaustion and have a sustained swimming speed of no more than 0.05 m/s for long periods (McCleave 1980). Glass eels resting on the seabed would be unlikely to be abstracted as the SZC intake surfaces would be 1.5 to 3.5m above the bed. The only times that glass eels would be at risk is when they were settling towards or moving off the seabed and then only for those that were within a worst case 7 m of the intakes (entrapment risk zone where velocities exceed 0.05 m/s). This represents a very small volume of water at the SZC intakes compared to the potential volume that the eels could settle in within Sizewell Bay and the abstraction risk is, therefore, considered minimal. The same argument would apply at SZB (whilst recognising that the risk would probably be larger due to the shallower water at the SZB intakes). Low entrapment potential combined with the expected low glass eel densities at the site and their migration pattern in surface waters would provide a coherent explanation of the absence of glass eels in the SZB entrainment monitoring surveys.

The targeted glass eel surveys only detected 1 individual from which it is not possible to deduce anything about their spatial distribution in Sizewell Bay and, in particular, whether the expected density would be lower or higher at the SZC intakes than at the SZB intakes. However, it is known that glass eels are seeking a freshwater signal as a cue to migrate up estuary. Due to dilution and the effects of tidal advection, the probability of detecting such signals will reduce rapidly with distance from the coast especially given the very strong shore parallel tidal currents along the Suffolk coast and the presence of offshore sandbanks. Many of the freshwater discharges on the East Anglian coast are relatively small, especially in the vicinity of Sizewell e.g. the Blyth and discharges from the Minsmere sluice, and such small signals in combination with the effects of dilution and tidal advection would indicate that a close to shore migration strategy would be the most likely to be allow the eels to find estuaries. On that basis the working hypothesis is that glass eels migrating on the coast will preferentially swim close to the coast and that their density offshore at the location of the SZC intakes would be lower than at the SZB intakes. Some evidential support for this hypothesis is provided by glass eel behaviour in lower estuaries where it is known that they occur in the highest densities closest to the shore when migrating up the estuary (for example in the Severn Estuary, BEEMS Technical Report TR274).

Considering the totality of the monitoring evidence and the implications of glass eel migration pattern around the UK, it is considered that the conclusions in BEEMS Technical Report TR318 that the density of glass eels off Sizewell was very low and that the risk of any significant entrainment effects on glass eel recruitment would be negligible is supported by the evidence.

Yellow eels

SZB impingement monitoring during the CIMP programme (with 10 mm mesh filtration) detected 2 glass eels (67.5 mm long) and a number of yellow eels ranging in length from 228 mm to 893 mm (

Figure 7 and Figure 8) i.e. with body widths from 14.25 mm to 55.8 mm (using morphological data reported in Environment Agency 2005). Ninety percent of the impinged eels were greater than 280 mm in length with a median length of approximately 400 mm. The length distribution of yellow eels is similar to that obtained from impingement sampling at Hinkley Point B. Studies to determine the age of the eels caught at Sizewell have not been undertaken but assuming a similar growth curve to that found at Oldbury in the Severn Estuary (Bird *et al* 2008), the yellow eels impinged at Sizewell B ranged from 2 to >25 years old. Yellow eels were caught throughout the year with the peak period of impingement in October and November and lowest catches in February to April and in December (Figure 9). From the length data and the SZB 10 mm mesh size it can reasonably be hypothesised that small (i.e. very young) yellow eels and elvers were not present at Sizewell; if young yellow eels had been present the length distribution would have been expected to continue down to below 160mm. Similarly, elvers were unlikely to have been present in any significant numbers as these larger fish would have been more likely to have been impinged than the two glass eels and would be unlikely to be present without young yellow eels. No glass eels or elvers were found in entrainment sampling. From these data it was concluded that yellow eels above 228 mm will be at risk of impingement at SZC with the majority of fish at risk having a length greater than 280 mm. All of the yellow eels are expected to be able to pass through the proposed 75 mm trash bar spacing at SZC.

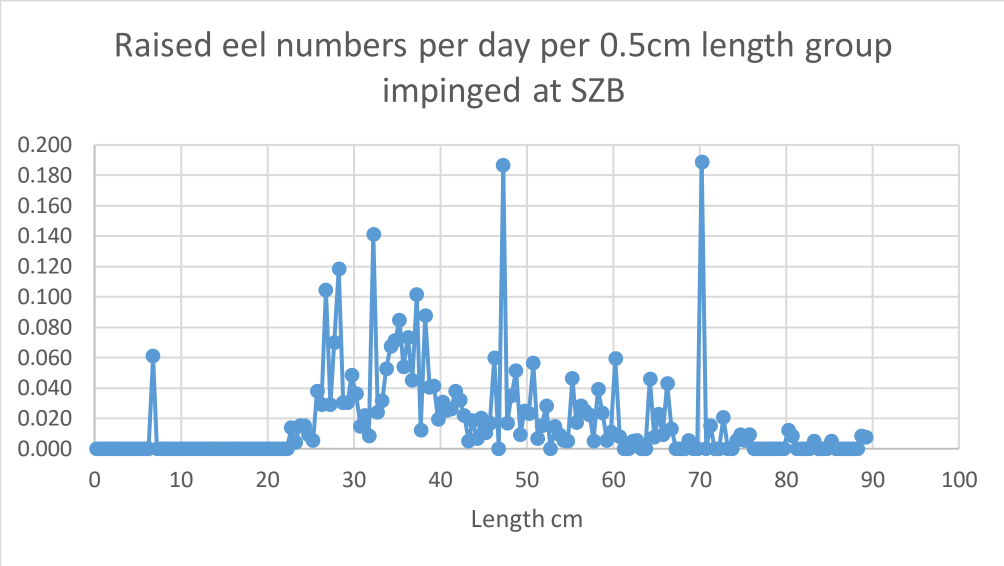


Figure 7 SZB impinged eel length frequency 2009-2017. The peak at 6.75cm corresponds to the 2 glass eels that were impinged in the period.

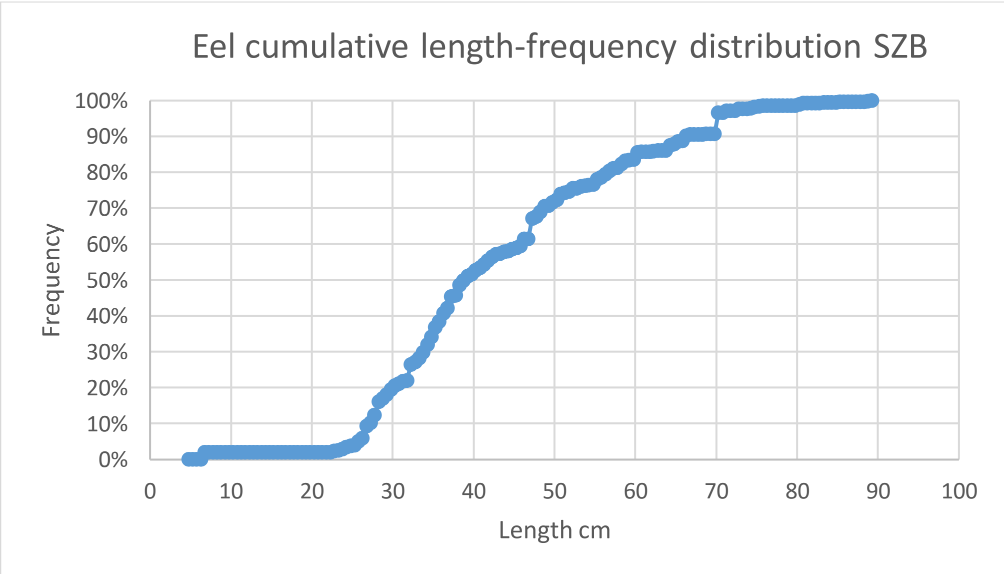


Figure 8 SZB impinged eel cumulative length frequency distribution 2009-2017.

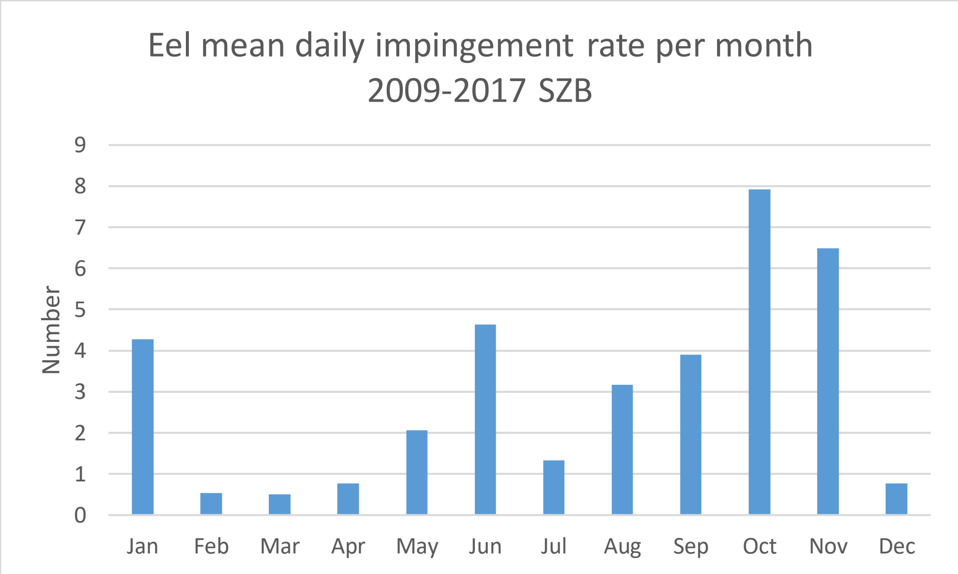


Figure 9 Variation in mean impingement rates by month at SZB.

Silver eels

No silver eels were caught in the SZB CIMP programme but this is not surprising as this life stage is known to migrate near to the surface at night and would be at a low risk of impingement at SZB's seabed mounted intakes and much less at SZC due to the deeper intakes.

7.6.4.4 Assessment of predicted effects of SZC

Although comparisons of eel mortalities due to impingement with population estimates for individual catchments are theoretically possible, there is uncertainty as to which are the relevant populations, and the European eel is considered to be a single reproductive stock throughout its distribution range. Given the small scale of the yellow and silver eel fisheries along the Suffolk coast, the most appropriate indicator of the perceived impact of the Sizewell power station on local eel stocks is considered to be a comparison between impingement data for eels by life stage, (raised to an equivalent silver eel biomass assuming 90 g for males, 570 g for females and a 1:1 sex ratio Dr. A. Walker, Cefas, pers. comm.) and, for fisheries, the combined mean yellow and silver eel catch for 2010-2017 (13.9 t) and, for the population, the mean estimated silver eel production for the Anglian RBD (78.6 t).

Based on the scaled-up CIMP dataset and assuming that the proposed LVSE intake head design and the FRR were fitted, the total annual predicted impingement of eel at a SZC, would be about 356 fish. Using a length-weight conversion factor of 0.329 kg per fish derived assuming a 50:50 sex ratio, that males mature at 89.9g and that females mature at 568.9g (Aprahamian, 1988), 356 eels equates to 0.12 t, equivalent to 0.15 % of the estimated RBD biomass. This latter figure is an overestimate as due to the lack of necessary biological and population data, it has not been possible to date to derive an EAV for eel, so a worst-case value of 1 has been assumed. Based on the eye index (Beullens *et al*., 1997) of biologically sampled impinged individuals at SZB (n = 89), all eels impinged were yellow eels and would have had an EAV of < 1 due to the natural mortality experienced by yellow eels in their many year growth period before maturation to silver eels. For example, the natural mortality of yellow eels is estimated to be 13% per annum, Dekker 2000. The yellow eels abstracted at SZB were in the age range 2 to >25 years and the majority would have remained at the yellow stage for many years before maturation. Assuming that the yellow eels would have spent an average of 5 years at Sizewell before maturation, the predicted SZC impingement effect would be reduced by 50% to 0.075% of the RBD silver eel biomass.

Another means of putting the SZC impingement estimate into context is to consider the predicted eel loss of 0.12 t per annum in the context of the estimated total EU anthropogenic impact on the stock of approximately 4,900 t per annum (Section 7.6.4.1). I.e. The SZC impact is equivalent to 0.002% of the total EU anthropogenic impact (from licenced fishing and secondly from hydropower and pumps operating largely in rivers).

As justified above and in BEEMS Technical Report TR318, the predicted effect of SZC entrainment on glass eel recruitment from SZC is considered negligible.

Compared to the other anthropogenic impacts on the stock, the predicted SZC entrapment effect on the EU component of the European eel stock is more than 4 orders of magnitude lower.

7.6.5 North Sea Herring and Blackwater Herring

The predicted effect of herring impingement in SZC is a negligible 0.01% SSB or 132 t per annum (Table 14).

The North Sea herring population has the following characteristics:

- North Sea Herring have multiple spawning grounds in the North Sea and eastern English Channel
- The different spawning populations are not genetically distinct.
- Scientific hypothesis: There is one population which is mixed during summer feeding and which migrates down the North Sea with subsets separating off to breed on route using broadly defined spawning areas as autumn, winter and finally spring spawners. The migration proceeds from Shetland to the eastern Channel and then northwards again (including some movement along the east Anglian coast from the Thames to the Humber)
- Different breeding times produce different growth patterns but the morphometric differences between fish are generally unreliable as an indicator of spawning type (autumn, winter, spring). Patterns in otolith rings are considered the most reliable indicator of spawning type.
- Blackwater herring are a spring spawning stock from February to April that spawn on the Eagle Bank at the entrance to the Blackwater Estuary in Essex and uniquely for North Sea herring have their own catch quota.

The main North Sea and Blackwater herring stocks are very different:

- The Blackwater stock is small with a management target for an SSB of at least 410 t.
- Historically the Blackwater stock was of little commercial interest due to the small size of the fish until the North Sea herring stock collapse that started in 1955 and the consequent reduction in landings. This stimulated commercial interest in the Blackwater stock that grew from 1958 with peak catches in 1972/73 of 600 t shortly followed by stock collapse and a complete closure of the fishery in 1979/80. The fishery reopened in 1980/81 under a management control regime designed to prevent a reoccurrence of overfishing.
- Recruitment to the Blackwater population has been poor in recent years and catches are now limited to 10 t per year for monitoring purposes from an SSB of approximately 200 t. An SSB of approximately 200 t is regarded as the minimum SSB to avoid stock collapse.
- The North Sea herring population has recovered to an SSB of greater than 2M tonnes but markets and fish landings are much lower than they were historically despite the substantial stock size.

Stakeholders have asked why herring impingement at Sizewell is considered to be from the main North Sea stock rather than the Blackwater stock.

As described above, the different stocks cannot be distinguished by genetics and so a weight of evidence approach has been used to ascribe origin to the herring impinged at Sizewell that considered:

- The timing of herring impingement at Sizewell compared with spawning in the Thames Estuary.

- Trends in SZB impingement rates compared with trends in the Blackwater SSB.
- The size of the SZB herring impingement and the implications if these fish had originated from the Blackwater stock.

Sizewell impingement is dominated by adults with numbers peaking in February and March and declining in April (BEEMS Technical Report TR345). Blackwater herring typically spawn at Eagle bank from the end February to mid-April (Fox 2001) and then return to southern North Sea feeding grounds. The coincident timing of spawning in the Thames Estuary and impingement at Sizewell indicates the fish caught at Sizewell are unlikely to be from the Blackwater. If Sizewell B was intercepting post spawning fish moving northwards, the peak catches would be expected in April or May with minimal catches in February when Blackwater herring are congregated in the Thames Estuary close to Eagle Bank.

The trend in impingement numbers at SZB has been increasing rapidly (Figure 10) which is contrary to the flat, possibly declining trend in the Blackwater SSB (Figure 11).

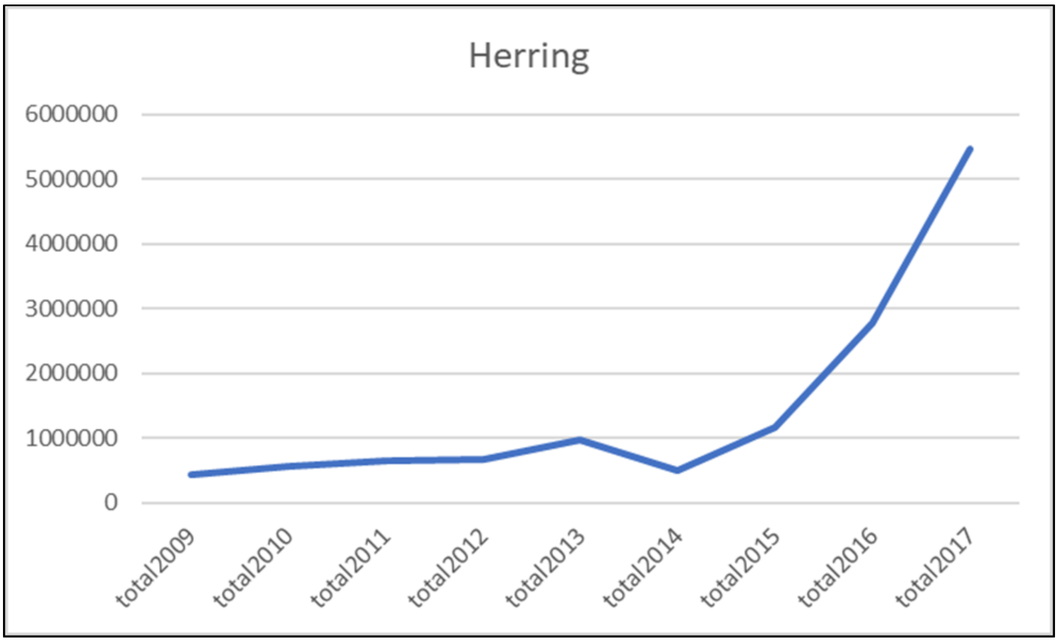


Figure 10 Herring impingement numbers at SZB (CIMP programme) by year.

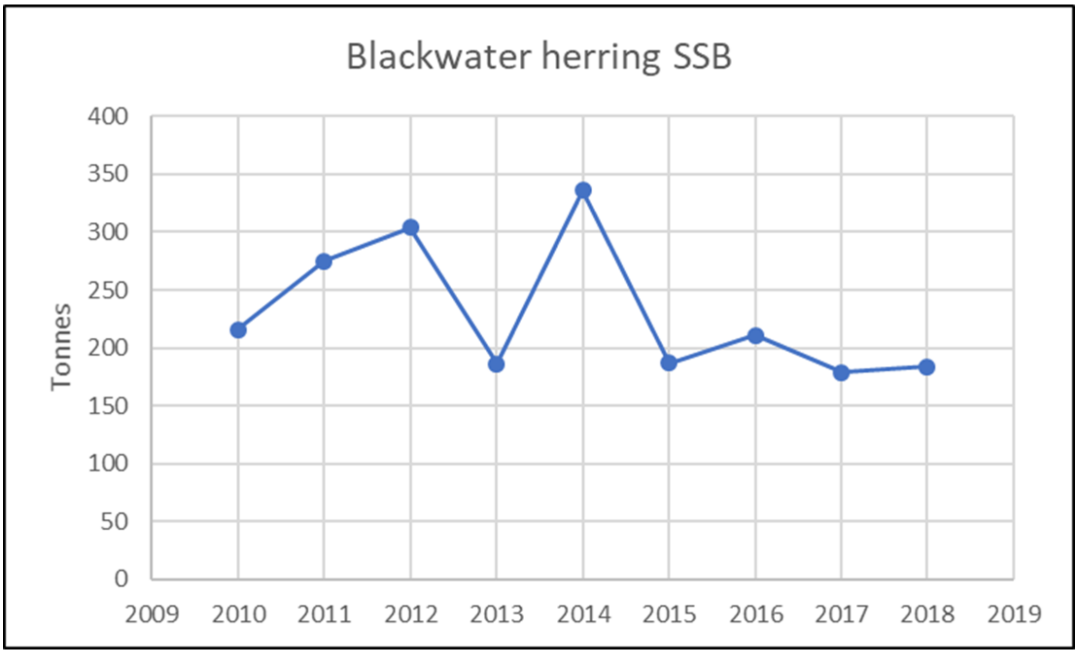


Figure 11 Change in the Blackwater herring SSB from 2010 to 2018 (Source Cefas).

Finally, the Blackwater SSB is less than or equal to 200 t at present (Figure 11). SZB annual mean herring impingement is estimated at 136 t (Appendix D2). If the SZB impingement was from the Blackwater herring stock that would represent an impingement impact of approximately 68% SSB that at the current SSB level would be completely unsustainable and lead to rapid total collapse of the stock to negligible levels and subsequently to minimal impingement at Sizewell. This has not happened, in fact SZB impingement numbers are increasing. This indicates that the SZB impingement is from the wider North Sea stock, not the Blackwater.

The weight of evidence therefore indicates that Sizewell impingement is from main North Sea stock (as assessed by ICES and used in this report) and not the Thames estuary Blackwater stock.

7.7 SZC predicted entrapment effects (impingement + entrainment)

In addition to the impingement of larger fish, invertebrates and other material, smaller organisms will also be abstracted by SZC but will not be retained on the drum screens. These smaller organisms will be entrained through the cooling water systems. The effects of entrainment on fish and other plankton is assessed in a separate report (BEEMS Technical Report TR318). Results indicated that apart from sand goby, the entrainment impacts for the key taxa present are significantly less than 1 %.

However, it is necessary to consider impingement and entrainment together, as a single entrapment impact for SZC. This is slightly complicated by the fact that the two data sets used for the respective assessments are different. For impingement the dataset spans 9 years, and the predictions are based on modelled mean monthly impingement values, using all year’s data. Impingement losses are considered against a mean SSB or landings value for the years 2009 – 2017 (the years of sampling). For entrainment, the predictions are based on a single year’s sampling (2010), and the losses were compared against the SSB and landings data for that year.

Entrapment has been estimated here by summing the % losses of impingement and entrainment. Although not a direct comparison of SSB or landings in a given year, given the extremely low entrainment losses of key taxa, when compared against SSB or landings, it would require annual changes in SSBs or landings of 2 or 3 orders of magnitude to significantly affect the combined total losses.

Combined impingement and entrainment estimated have been made for SZC:

a. without mitigation (Table 16);

b. with the proposed LVSE intake heads and FRR fitted (Table 17), and

c. With the impingement assessment with LVSE intakes and FRR systems fitted updated as detailed in section 7.5 for thin lipped grey mullet and bass (Table 18).

Results are similar to those obtained for impingement alone in that without mitigation only seabass, thin-lipped grey mullet, and European eel exceed 1 %. These species are joined by the sand goby for combined entrapment. With the proposed impingement mitigations, only sand goby exceeds 1 % of the stock comparator. However, sand goby is an unexploited short-lived stock and in such circumstances the appropriate comparator for negligible effects is with 10% of the SSB. As such the predicted entrapment of sand goby is negligible.

Table 16 Annual mean SZC entrapment predictions (impingement + entrainment) **with no impingement mitigation**. For impingement, losses have been converted to adult equivalent (EAV) numbers and weights (t) and calculated as a % of either the mean stock SSB (t) or mean international landings (t) for the period 2009-2017. For entrainment, the worst-case losses have been converted to EAV numbers and weight and calculated as a % of the SSB and landings in 2010 only. Species where the entrapment weight > 1 % of the relevant stock comparator (either SSB or landings – given in bold) are shaded red. Numbers in red font are either estimates of the population numbers (e.g. sand goby) or reported catch numbers (salmon & sea trout)

	Impingement						Entrainment						Entrapment			
Species	EAV number	EAV weight	Mean SSB	% of SSB	Mean Landings	% of landings	EAV number	EAV weight	SSB 2010	% of SSB	Landings 2010	% of landings	EAV number	EAV weight	% of SSB	% of landings
Sprat	5,352,978	56.23	220,757	0.03	151,322	0.04	199,715	2.00	225,041	0.00	143,500	0.00	5,552,693	58.23	**0.03**	0.04
Herring	1,827,944	344.87	2,198,449	0.02	400,244	0.09	23,992	4.18	2,023,720	0.00	187,600	0.00	1,851,936	349.05	**0.02**	0.09
Whiting	664,261	189.86	151,881	0.13	17,570	1.08	-	-	-	-	-	-	664,261	189.86	**0.13**	0.62
Bass	128,861	197.26	14,897	1.32	3,051	6.47	36	0.05	20,780	0.00	4,768	0.00	128,897	197.31	**1.32**	6.47
Sand goby	381,612	0.73	205,882,353	0.19	NA	NA	2,892,198	-	205,882,353	1.40			3,273,810		**1.59**	NA
Sole	53,233	11.40	43,770	0.03	12,800	0.09	631	0.14	31,358	0.00	12,603	0.00	53,864	11.54	**0.03**	0.09
Dab	66,211	2.70	NA	NA	6,135	0.04	21,810	0.87	NA	NA	8,279	0.01	88,021	3.57	NA	**0.05**
Anchovy	71,952	1.49	NA	NA	1,625	0.09	2,869	0.06	NA	NA	727	0.01	74,821	1.55	NA	**0.10**
Thin-lipped grey mullet	5,642	2.93	NA	NA	120	2.45	-	-	-	-	-	-	5,642	2.93	NA	**2.45**
Flounder	17,631	1.44	NA	NA	2,309	0.06	2	0.00	NA	NA	3,365	0.00	17,633	1.44	NA	**0.06**
Plaice	8,734	2.15	690,912	0.00	80,367	0.00	-	-	-	-	-	-	8,734	2.15	**0.00**	0.00
Smelt	18,170	0.30	105,733,825	0.02	8	3.56	-	-	-	-	-	-	18,170	0.30	**0.02**	3.56
Cod	6,049	15.74	103,025	0.02	34,701	0.05	-	-	-	-	-	-	6,049	15.74	**0.02**	0.03
Thornback ray	2,082	6.65	NA	NA	1,573	0.42	-	-	-	-	-	-	2,082	6.65	NA	**0.42**
River lamprey	6,720	0.53	62	0.86	1	47.65	-	-	-	-	-	-	6,720	0.53	**0.86**	47.65
Eel	4,516	1.49	79	1.89	14	10.70	-	-	-	-	-	-	4,516	1.49	**1.89**	10.70
Twaite shad	3,601	1.13	7,519,986	0.05	1	84.60	-	-	-	-	-	-	3,601	1.13	**0.05**	84.60
Horse mackerel	4,077	0.57	NA	NA	20,798	0.00	-	-	-	-	-	-	4,077	0.57	NA	**0.00**
Mackerel	628	0.20	3,888,854	0.00	1,026,828	0.00	-	-	-	-	-	-	628	0.20	**0.00**	0.00
Tope	64	0.44	NA	NA	498	0.09	-	-	-	-	-	-	64	0.44	NA	**0.09**
Sea trout	10	0.02	NA	NA	39,795	0.02	-	-	-	-	-	-	10	0.02	NA	**0.02**
Allis shad	5	0.00	27,397	0.02	0	1.79	-	-	-	-	-	-	5	0.00	**0.02**	1.79

Species	Impingement						Entrainment						Entrapment			
	EAV number	EAV weight	Mean SSB	% of SSB	Mean Landings	% of landings	EAV number	EAV weight	SSB 2010	% of SSB	Landings 2010	% of landings	EAV number	EAV weight	% of SSB	% of landings
Sea lamprey	5	0.01	NA	NA	NA	NA	-	-	-	-	-	-	5	0.01	NA	NA
Salmon	0	0.00	NA	NA	38,456	0.00	-	-	-	-	-	-	0	0.00	NA	**0.00**

Table 17 Annual mean SZC entrapment predictions (impingement + entrainment) **considering the effect of the intake head design and with FRR systems fitted**. Losses have been converted to adult equivalent (EAV) numbers and weights (t) and calculated as a % of either the mean stock SSB (t) or mean international landings (t). Species where the impingement weight > 1 % of the relevant stock comparator (either SSB or landings – given in bold) are shaded red. Numbers in red font are either estimates of the population numbers (e.g. sand goby) or reported catch numbers (salmon & sea trout)

Species	Impingement						Entrainment						Entrapment			
	EAV number	EAV weight	Mean SSB	% of SSB	Mean landings	% of landings	EAV number	EAV weight	SSB 2010	% of SSB	Landings 2010	% of landings	EAV number	EAV weight	% of SSB	% of Landings
Sprat	2,050,190	21.53	220,757	0.01	151,322	0.01	199,715	2	225,041	0.00	143,500	0.00	2,249,905	23.53	**0.01**	0.01
Herring	700,103	132.08	2,198,449	0.01	400,244	0.03	23,992	4	2,023,720	0.00	187,600	0.00	724,095	136.26	**0.01**	0.03
Whiting	140,044	40.03	151,881	0.03	17,570	0.23	-	-	-	-	-	-	140,044	40.03	**0.03**	0.13
Bass	27,172	41.6	14,897	0.28	3,051	1.36	36	0	20,780	0.00	4,768	0.00	27,208	41.65	**0.28**	1.36
Sand goby	30,108	0.06	205,882,353	0.01	NA	NA	2,892,198	-	205,882,353	1.40	0	0.00	2,922,306		**1.42**	0.00
Sole	4,200	0.9	43,770	0	12,800	0.01	631	0	31,358	000	12,603	0.00	4,831	1.04	**0.00**	0.01
Dab	13,656	0.56	NA	NA	6,135	0.01	21,810	1	NA	NA	8,279	0.00	35,466	1.43	NA	**0.02**
Anchovy	27,558	0.57	NA	NA	1,625	0.04	2,869	0	NA	NA	727	0.00	30,427	0.63	NA	**0.05**
Thin-lipped grey mullet	1,190	0.62	NA	NA	120	0.52	-	-	-	-	-	-	1,190	0.62	NA	**0.52**
Flounder	1,559	0.13	NA	NA	2,309	0.01	2	0	NA	NA	3,365	0.00	1,561	0.13	NA	**0.01**
Plaice	689	0.17	690,912	0	80,367	0	-	-	-	-	-	-	689	0.17	**0.00**	0.00
Smelt	6,959	0.12	105,733,825	0.01	8	1.36	-	-	-	-	-	-	6,959	0.12	**0.01**	1.36
Cod	1,395	3.63	103,025	0	34,701	0.01	-	-	-	-	-	-	1,395	3.63	**0.00**	0.01
Thornback ray	164	0.52	NA	NA	1,573	0.03	-	-	-	-	-	-	164	0.52	NA	**0.03**
River lamprey	530	0.04	62	0.07	1	3.76	-	-	-	-	-	-	530	0.04	**0.07**	3.76
Eel	356	0.12	79	0.15	14	0.84	-	-	-	-	-	-	356	0.12	**0.15**	0.84
Twaite shad	1,379	0.43	7,519,986	0.02	1	32.40	-	-	-	-	-	-	1,379	0.43	**0.02**	32.40
Horse mackerel	1,561	0.22	NA	NA	21,442	0	-	-	-	-	-	-	1,561	0.22	NA	**0.00**
Mackerel	241	0.08	3,888,854	0	1,026,828	0	-	-	-	-	-	-	241	0.08	**0.00**	0.00
Tope	5	0.03	NA	NA	498	0.01	-	-	-	-	-	-	5	0.03	NA	**0.01**
Sea trout	4	0.01	NA	NA	39,795	0.01	-	-	-	-	-	-	4	0.01	NA	**0.01**
Allis shad	2	0	27,397	0.01	0	0.68	-	-	-	-	-	-	2	0.00	**0.01**	0.68
Sea lamprey	0	0	NA	NA	NA	NA	-	-	-	-	-	-	0	0.00	NA	NA

Species	Impingement						Entrainment						Entrapment			
	EAV number	EAV weight	Mean SSB	% of SSB	Mean landings	% of landings	EAV number	EAV weight	SSB 2010	% of SSB	Landings 2010	% of landings	EAV number	EAV weight	% of SSB	% of Landings
Salmon	0	0	NA	NA	38,456	0	-	-	-	-	-	-	0	0.00	NA	**0.00**

Table 18 Annual mean SZC entrapment predictions (impingement + entrainment) **considering the effect of LVSE intake heads and FRR systems fitted and the corrections to thin lipped grey mullet and bass impingement assessment detailed in Section 7.5 (main changes shown in yellow)**. Losses have been converted to adult equivalent (EAV) numbers and weights (t) and calculated as a % of either the mean stock SSB (t) or mean international landings (t). Species where the impingement weight > 1 % of the relevant stock comparator (either SSB or landings – given in bold) would be shaded red (there are none in this Table) with the exception of sand goby where a 10% of SSB or landings comparator has been used. Numbers in red font are either estimates of the population numbers (e.g. sand goby) or reported catch numbers (salmon & sea trout)

	Impingement						Entrainment						Entrapment			
Species	EAV number	EAV weight	Mean SSB	% of SSB	Mean landings	% of landings	EAV number	EAV weight	SSB 2010	% of SSB	Landings 2010	% of landings	EAV number	EAV weight	% of SSB	% of Landings
Sprat	2,050,190	21.53	220,757	0.01	151,322	0.01	199,715	2	225,041	0.00	143,500	0.00	2,249,905	23.53	**0.01**	0.01
Herring	700,103	132.08	2,198,449	0.01	400,244	0.03	23,992	4	2,023,720	0.00	187,600	0.00	724,095	136.26	**0.01**	0.03
Whiting	140,044	40.03	151,881	0.03	17,570	0.23	-	-	-	-	-	-	140,044	40.03	**0.03**	0.13
Bass	2,717	4.16	14,897	0.03	3,051	0.14	36	0	20,780	0.00	4,768	0.00	2,753	4.16	**0.03**	0.14
Sand goby	30,108	0.06	205,882,353	0.01	NA	NA	2,892,198	-	205,882,353	1.40		0.00	2,922,306		**1.41**	0.00
Sole	4,200	0.9	43,770	0	12,800	0.01	631	0	31,358	0.00	12,603	0.00	4,831	1.04	**0.00**	0.01
Dab	13,656	0.56	NA	NA	6,135	0.01	21,810	1	NA	NA	8,279	0.00	35,466	1.43	NA	**0.02**
Anchovy	27,558	0.57	NA	NA	1,625	0.04	2,869	0	NA	NA	727	0.00	30,427	0.63	NA	**0.05**
Thin-lipped grey mullet	1,190	0.62	600	0.10	120	0.52	-	-	-	-	-	-	1,190	0.62	**0.10**	0.52
Flounder	1,559	0.13	NA	NA	2,309	0.01	2	0	NA	NA	3,365	0.00	1,561	0.13	NA	**0.01**
Plaice	689	0.17	690,912	0	80,367	0.00	-	-	-	-	-	-	689	0.17	**0.00**	0.00
Smelt	6,959	0.12	105,733,825	0.01	8	1.36	-	-	-	-	-	-	6,959	0.12	**0.01**	1.36
Cod	1,395	3.63	103,025	0.00	34,701	0.01	-	-	-	-	-	-	1,395	3.63	**0.00**	0.01
Thornback ray	164	0.52	NA	NA	1,573	0.03	-	-	-	-	-	-	164	0.52	NA	**0.03**
River lamprey	530	0.04	62	0.07	1	3.76	-	-	-	-	-	-	530	0.04	**0.07**	3.76
Eel	356	0.12	79	0.15	14	0.84	-	-	-	-	-	-	356	0.12	**0.15**	0.84
Twaite shad	1,379	0.43	7,519,986	0.02	1	32.40	-	-	-	-	-	-	1,379	0.43	**0.02**	32.40
Horse mackerel	1,561	0.22	NA	NA	20,798	0.00	-	-	-	-	-	-	1,561	0.22	NA	**0.00**
Mackerel	241	0.08	3,888,854	0	1,026,828	0.00	-	-	-	-	-	-	241	0.08	**0.00**	0.00
Tope	5	0.03	NA	NA	498	0.01	-	-	-	-	-	-	5	0.03	NA	**0.01**

Species	Impingement						Entrainment						Entrapment			
	EAV number	EAV weight	Mean SSB	% of SSB	Mean landings	% of landings	EAV number	EAV weight	SSB 2010	% of SSB	Landings 2010	% of landings	EAV number	EAV weight	% of SSB	% of Landings
Sea trout	4	0.01	NA	NA	39,795	0.01	-	-	-	-	-	-	4	0.01	NA	**0.01**
Allis shad	2	0	27,397	0.01	0	0.68	-	-	-	-	-	-	2	0.00	**0.01**	0.68
Sea lamprey	0	0	NA	NA	NA	NA	-	-	-	-	-	-	0	0.00	NA	NA
Salmon	0	0	NA	NA	38,456	0	-	-	-	-	-	-	0	0.00	NA	**0.00**

7.8 Consideration of potential local effects on the fish assemblage at Sizewell

In Section 7.7 the predicted SZC entrapment effect on fish populations has been assessed against ICES derived spawning stock biomasses (SSB) (or international landings as a highly precautionary proxy for SSB). Comparison against SSBs is the internationally recognised best practice way that the much larger effects of fishing at either a fleet or individual boat level are assessed.

SZC Stakeholders have indicated that in principle they agree with this assessment methodology, but questions have been raised on whether SZC will have any localised entrapment effects on fish. This section of the assessment considers that question in detail.

Section 5.10 describes how ICES decides on the definition of stock units based upon a mature weighting of the best available scientific evidence. As a result of decades of research, it is clear that the population structures of marine species fall along a continuum from panmictic (e.g. European eel, *Anguilla anguilla*) to numerous distinct sub-populations (e.g. North Sea herring, *Clupea harengus*) with the majority of species exhibiting complex structure.   In the open sea, the sub-populations of many species mix to a considerable extent; especially during summer feeding and on nursery areas, with harvesting affecting multiple components of the overall population simultaneously. ICES' definition of stock units integrates all of the information on site fidelity to spawning, nursery and feeding areas together with knowledge of migration patterns and the degree of intermixing that takes place between any sub populations. Stock units are not static and change when the weight of evidence indicates that a change would be likely to lead to better assessments and management advice. At the request of SZC stakeholders, the specific status of the large bass stock area used in the SZC entrapment assessment was considered (Section 5.10.1). The current stock unit comprises ICES Divisions 4b-c, 7a and 7d-h (central and southern North Sea, Irish Sea, English Channel, Bristol Channel, and Celtic Sea) and SZC stakeholders have queried whether that area is appropriate based upon known bass site fidelities seasonally and at different life stages. The ICES bass working group is fully aware of this research which was weighed together with the extensive monitoring by EU member states (e.g. 27 bass nursey sites are monitored in the UK alone, Source Cefas) and the stock unit definition takes account of site fidelity and the degree of mixing between the sub populations. The stock unit was last reviewed in 2018 and found to fully appropriate based upon the evidence. Coordinated research is currently underway on bass stock identity, but the direction of that research may lead to a larger stock unit area, not a smaller one (Section 5.10.2).

In section 5.10.3 we concluded that ICES' stock boundaries are based on a mature weighing of all of the best scientific evidence available and considering the negligible predicted SZC impacts compared to those of fishing (Table 21 and Table 22, and the precautionary nature of ICES' estimates of SSBs, we could find no justification not to use the ICES' stock definitions to assess SZC effects on fish,

As explained previously, where SSB data do not exist the effects of SZC are compared with international landings from the stock unit which provides a highly precautionary estimate of effects as landings are less, normally much less, than stock sizes.

For some data poor stocks, there is no assessment of the SSB for the stock and in those cases the SZC assessment has, on a precautionary basis, assessed SZC effects by comparison against a part of the stock where only partial data are available. The effects assessment in those cases provides a precautionary overestimate of SZC effects (In this report such an assessment of SZC effects against partial stock estimates has been undertaken for European eel, twaite shad, allis shad, cucumber smelt and river lamprey).

7.8.1 Fish at Sizewell in a southern North Sea context

The fish species at Sizewell live and move in an unconstrained coastal environment with most species undertaking wide spatial migrations throughout the year; in particular migrating fish are not forced to pass close to power station intakes as could be the case in a narrow river or estuary. In the coastal environment at Sizewell any local reduction in fish numbers by SZC would be expected to be replaced by incoming species competing for habitat and food resources. Unlike in some estuarine environments, there are no unique features or resources that fish populations are dependent upon in the vicinity. For example, estuaries often function as nursery areas for young fish but the Sizewell area does not have an extensive nursery role for most species (see Section 7.8.2) as exemplified by the older and larger fish found at the site compared with an estuary such a Hinkley Point (Table 19). From Table 19 it can be seen that for the typical species listed, only Thornback Ray are larger at Hinkley Point than at Sizewell and for herring, whiting and cod, in particular, Sizewell has much lower numbers of 0 group (less than 1 year old) fish. (The lower the EAV in the range 0 to 1, the greater the proportion of juvenile fish with EAVs of around 0.1 or below consisting predominantly of 0 or possibly 1 group fish).

The fish assemblage at Sizewell is a reflection of the seasonal migrations of fish into and out of the area and fish lost at Sizewell C would be rapidly replaced by exchanges with populations from the wider southern North Sea.

Table 19 Differences in age and maturity of fish caught at Sizewell and Hinkley Point as reflected in Equivalent Adult Values (EAV) where an EAV of 1 is a mature adult.

Species	SZ EAV	HP EAV
Sprat	0.751	0.556
Herring	0.715	**0.113**
Whiting	0.356	**0.098**
Bass	0.224	0.121
Dover sole	0.213	0.236
Cod	0.359	**0.018**
Thornback Ray	0.193	0.339
Plaice	0.345	0.192

Note: EAVs are from BEEMS Technical Reports TR383 (Sizewell) and TR426 (Hinkley Point) respectively.

7.8.2 What local effects of SZC entrapment might be important?

If present, the following potential local effects could be important:

i. **If a reduction in local fish numbers adversely affected a spawning or nursery habitat in the vicinity of Sizewell that was critical to the sustainability of a stock.**
Limited local spawning does take place in the region of Sizewell, predominantly by Dover Sole and Anchovy (the two species contributed a total of 85% of measured egg numbers at Sizewell), but the measured egg density indicates that the area is not important to the species which are geographically widely distributed (BEEMS Technical Report TR318). Similarly, the Sizewell region provides some nursery habitat predominantly for gobies and sprat (86% of measured larval and juvenile fish numbers) but again the measured densities indicate that the area is of low importance for these two widely distributed species (BEEMS Technical Report TR318). For all of the species predicted to be entrained by SZC the predicted stock level effects are negligible (BEEMS Technical Report TR318). The importance of the Sizewell Bay area for fish spawning and nursery habitat is, therefore, low and there are much more important habitat areas over the southern North Sea and beyond (BEEMS Technical Report TR345).

ii. **If a reduction in local fish numbers adversely affected prey availability for predators**.
The local ecosystem is founded upon predator-prey relationships and so localised depletion of a prey resource could adversely affect the predator sustainability e.g. HRA protected marine birds predating on fish. Such food webs are routinely subject to high levels of natural variability and predators have evolved adaptation strategies to cope including prey species switching and changes to foraging behaviour. Predators will only be sensitive to large and sustained changes to prey

abundance unless their foraging range is small and their ability to change their foraging range is limited. Of the protected marine birds in the vicinity of Sizewell, little tern during the breeding season have by far the smallest foraging range (BEEMS Technical Report TR431) and could therefore be vulnerable to any localised depletion of fish prey due to SZC. Breeding little tern are designated in the Minsmere-Walberswick SPA however colony locations for this species are known to be highly variable with time. At classification in 1991, the SPA's breeding population was 28 breeding pairs however, since classification, the numbers of little tern using Minsmere-Walberswick SPA has decreased by approximately 95% to 1.6 breeding pairs (5 year mean peak count 2014-2018) (Natural England 2019).

The diet of breeding little tern at Sizewell is expected to consist of small schooling pelagic fish species that are found close to the sea surface and demersal fish in the shallows such as gobies. During the breeding season little tern forage close to their colonies out to a maximum distance of approximately 2.4 km offshore (TR431). They would not, therefore, be expected to be foraging offshore of the Sizewell- Dunwich Bank in the vicinity of the proposed SZC intakes at approximately 3km offshore and would instead be foraging inshore i.e. within the zone of abstraction impact from SZB. Impingement data, indicates that sprat and herring are the most common pelagic fish in Sizewell coastal waters (TR431). The predicted impingement mortality from SZB is 22 t of sprat and 135 t of herring per annum (Appendix D2). All abstracted fish are returned to sea from SZB but these pelagic species are not expected to survive impingement. Assuming sprat and/or herring accounted for the whole diet of little terns, the calculated biomass required to sustain 28 pairs (with up to 3 chicks per pair) during the 4-month breeding season is less than 650 kg per annum.

The predicted effect of the SZB losses on the recognised stocks of sprat and herring are negligible (Appendix D2). If SZB was having a significant effect on the local sprat and herring abundance this would be apparent in the impingement numbers (which would collapse). However, the herring population at Sizewell is increasing (Figure 10) and the sprat population is the largest of any fish population at Sizewell and shows no discernible trend. Acoustic survey results show no localised reduction of pelagic shoaling fish in the vicinity of the SZB intake (BEEMS Technical Report TR359) and it is apparent that localised losses at SZB are being replaced by the constantly moving shoals from the wider North Sea. Once SZC is operational the same pattern is expected with no discernible differences in pelagic fish abundance in the vicinity of the SZC intakes. However, fish density at SZC would be immaterial to little tern breeding success as the abstraction zone would be too far offshore for the area to be important to little tern foraging.

7.8.3 SZC Impingement Risk Zones and the potential for very local fish depletion

Fish are only likely to be impinged if they move into the zone of influence of the intake heads. For SZC, a tidally averaged worst case of approximately 0.7m from the intake faces would experience an intake velocity of 0.3 m/s (BEEMS Scientific Position Paper SPP099). Compared with the spatial domain that the migrating fish move within, this abstraction risk zone is very small. At first site this could be considered as the zone where localised effects would be observed. However, due to the continual replacements by fish from a wider area there is no such extremely local effect as demonstrated by the impingement time series data.

7.8.4 Evidence of localised impingement effects from other sites

When the Hinkley Point A (HPA) station closed down a seawater abstraction of 44 cumecs was removed from the Hinkley Point intake structure. If an impingement impact of the size of the HPA abstraction was having any effect on local fish populations then the closure should have been detectable in the 35+ years of the HPB impingement record. In practice no such effect could be detected, and it was concluded that the local fish assemblage was not sensitive to at least a 44 cumec reduction in unmitigated impingement pressure, BEEMS Technical Report TR456 (Note: HPA and HBP were not fitted with impingement mitigation measures). SZC would be fitted with the latest impingement mitigation technology e.g. LVSE intakes and an FRR system and to compare the impingement pressure exerted by the proposed station it is necessary to calculate an equivalent unmitigated abstraction (Table 20). For SZC this impact varies between 10.4 cumecs and 50.6 cumecs which for most species is less than the change in impingement of 44 cumecs that caused no detectable effect at Hinkley Point.

Table 20 SZC equivalent abstraction after impingement mitigation of LVSE intakes and FRR system in comparison with an unmitigated SZC abstracting 132 cumecs

Species group	Equivalent SZC abstraction in cumecs compared with an unmitigated station abstracting 132 cumecs
Pelagic	50.6
Demersal	27.8 – 30.3
Epibenthic	10.4

The only species that would experience an increase in impingement pressure that was greater than 44 cumecs would be the pelagic species (sprat, herring, anchovy, smelt, mackerel, horse mackerel, twaite shad and allis shad) where there would be a marginal increase in impingement pressure of 6.6 cumecs above the 44 cumec pressure that caused no detectable effect at Hinkley Point. Of these species only sprat and herring and to a much lower extent anchovy play an important role in the local food web and as discussed above they are widely distributed species that migrate continuously over very large spatial areas. The other 5 species (smelt, mackerel, horse mackerel, twaite shad and allis shad) are not abundant enough at Sizewell to be important to the local food web with only smelt having a non-trivial presence (Table 18).

The additional abstraction from SZC is not considered large enough to have any significant effect on pelagic fish numbers because, as explained above, SZC losses will be continuously replaced by pelagic fish moving into the local area from the wider southern North Sea.

7.8.5 SZC entrapment effects on smelt in the Alde Ore and potentially in the Blyth water bodies

Stakeholders have specifically asked whether SZC entrapment would significantly affect the abundance of smelt in the Alde Ore and possibly the Blyth water bodies.

The key facts about smelt in the vicinity of Sizewell are as follows:

i. Smelt are relatively common on the East Anglian coast. Comparative genomic analyses (BEEMS Technical Report TR423) have concluded that smelt from Sizewell and the Rivers Thames, Waveney, and Great Ouse are genetically homogeneous with no genetic structuring seen within the region indicating a single stock unit from at least the Greater Ouse to the Thames. Based upon fishing surveys it is considered likely that there is also a single population along the European coast from at least the Elbe to the Scheldt. The extent of mixing between the UK and European populations is currently unknown but based upon the comparative distances between the great Ouse and the Thames and Sizewell and the European coast the hypothesis is that there is a single southern North Sea population that mixes during summer feeding. This hypothesis is considered reasonable, but it is recognised that it has not yet been proven and appropriate samples for further genomic analyses are being acquired. The Elbe and the Scheldt have very large breeding populations of smelt and it is likely that they have extensive summer feeding grounds in the southern North Sea that may overlap areas used by UK sub populations.
ii. No smelt eggs or larvae are entrained at SZB as would expected given the known lifecycle of the species (Section 7.6.1).
iii. A breeding population in the Blyth is considered unlikely due to the lack of habitat and barriers to migration (Section 7.6.1).
iv. SZB does not impinge any 0-group smelt and predominantly catches 1yr old and older fish in the summer i.e. after the spring spawning period whilst the sub populations of the species are mixed on summer feeding grounds. Low numbers of fish are caught in the period early February to end April when mature adults would be spawning in rivers (BEEMS Technical Report TR345). SZB is not,

therefore, considered to be having any significant effects on smelt migrating to and/or from estuaries.

v. Smelt impingement numbers at SZB show no trend over the period 2009-2017 and numbers in that period are similar, possibly larger than in 1981/82. There is, therefore, no evidence that effects of fishing and anthropogenic mortality on the stock are unsustainable.

vi. Assessments of SZC entrapment effects compared with both UK landings (i.e. assuming an Anglian stock unit) or assuming a southern North Sea stock unit are both negligible (Section 7.6.1).

The weight of evidence is, therefore, that SZC entrapment will have no significant effect on any sub population of smelt in the Alde Ore. The Blyth is not considered to possess a spawning sub population due to a lack of suitable habitat.

7.8.6 Conclusions on potential local effects from SZC entrapment

An assessment of potential localised effects of SZC entrapment was undertaken and found no likely significant adverse effects on:

i. spawning or nursery areas in the vicinity of Sizewell
ii. the prey of HRA protected breeding little tern (the potentially most vulnerable species to localised effects on prey fish abundance at Sizewell)

7.9 Contextualising SZC entrapment losses

To place the predicted fish losses due to entrapment by SZC (Table 18) into context two analyses are presented:

i. Table 21 shows a comparison for those stocks where data are available between mean fishery landings as a percentage of SSB and SZC entrapment as a percentage of SSB for the period 2009-2017.

ii. Table 22 shows discarded fish weight as a percentage of landed weight by year for commercially exploited species compared with ICES records of fishery discards. The same table also shows predicted mean SZC entrapment weights as a percentage of the mean landings for each species.

Table 21 Comparison of mean fishery landings as a percentage of SSB with predicted SZC mean entrapment as a percentage of SSB for the period 2009-2017.

Species	Fisheries Landings as % SSB	SZC entrapment losses as % of SSB	Notes
Sprat	69%	0.01%	
Herring	18%	0.01%	Commercial market reduced after stock collapse and subsequent recovery
Whiting	20%	0.03%	
Bass	20%	0.03%	
Sole	29%	0.00%	
Plaice	12%	0.00%	
Cod	45%	0.00%	
Mackerel	26%	0.00%	
River lamprey	2%	0.07%	SSB is of Derwent population only. Landings and effort are restricted by licences.
Eel	18%	0.15%[1]	SSB is of Anglian RBD only.

[1] SZC effect overestimated due to use of incorrect EAV.

Table 21 shows that the fishery impact on each stock is much greater than that of SZC entrapment, often by orders of magnitude.

Table 22 Discards by year as a percentage of landed fish weight compared with predicted SZC entrapment as a percentage of landed fish weight.

Year	Cod	Sole	Plaice	Dab	Whiting	Flounder	Horse mackerel	Mackerel
2008	93.2	1.9	44.4	316.7	57.4	44.9	-	3.9
2009	62.8	4.3	39.5	474.7	54.5	47.9	-	3.6
2010	33.6	8.3	35.5	513.2	73.5	97.5	-	0.4
2011	27.2	7.2	33.6	599.0	66.1	55.7	-	0.6
2012	26.9	9.0	44.2	746.9	73.6	55.0	-	0.5
2013	33.8	11.2	25.2	808.9	46.4	80.9	-	0.1
2014	30.4	6.5	39.6	1064.7	66.6	59.1	-	0.1
2015	33.5	6.9	35.1	932.2	109.8	69.7	20.0	0.0
2016	32.0	5.4	30.4	881.2	107.2	35.9	11.1	0.2
2017	23.1	5.8	31.5	875.9	91.0	47.3	9.1	0.1
SZC Entrapment (% of landed weight)	0.01	0.01	0.00	0.02	0.23	0.01	0.00	0.00

For commercially important species, SZC entrapment losses are lower than <1% of landings. Discards as a percentage of landings weights vary dramatically depending on the species. However, entrapment losses are at least two orders of magnitude lower than the proportion of landed fish that is discarded annually. Expressing the results as weights can provide an even more compelling illustration and for example, the mean weight of cod discarded between 2009 and 2017 was 12,980 t whereas the predicted mean SZC entrapment loss for the same period was 3.6 t.

8 Shellfish impingement predictions for SZC

Four shellfish species (brown crab *Cancer pagurus*, European lobster *Homarus gammarus*, brown shrimp *Crangon crangon* and whelk *Buccinum undatum*) were defined as key benthic species on the basis of their socio-economic importance (all four species) and their ecological importance (brown shrimp only) (BEEMS Technical Report TR348). Of these, whelk were absent from the impingement dataset. The impact of SZC on whelks is therefore considered negligible and the species will not be considered further. Estimates of SZB and predictions of SZC impingement were made using the same methods described for the finfish species.

8.1 FRR system mortality

8.1.1 Trash rack mortality

As with finfish, the proportion of the shellfish species that would pass through the 75 mm trash rack was assessed (Table 23). For brown crab, carapace width (CW, mm) measurements were made on crabs sampled at the SZB site between 2014 and 2017 and used to calculate annual length distributions. The maximum width observed was > 75 mm, and the proportion that is likely to pass was calculated in a similar manner to finfish species. Only four lobsters were recorded in the CIMP dataset between 2009 and 2017, all of them prior to 2014. Only weights were recorded, and these were used to estimate the proportion that would pass through a 75 mm bar spacing.

Table 23 Proportion of shellfish, by species that will not pass through the 75 mm wide trash racks

Species	Calculation type	Size of largest SZB individual (mm)	Proportion not passing through trash rack	Comment
Brown shrimp	Group 1	Not measured	0.000	All shrimp will pass
Brown crab	Group 3	162	0.014	Calculate length at 75 mm width
Lobster		Not measured	0.500	Best estimate based on individual weights

8.1.2 FRR survival

According to Environment Agency (2005), crustaceans will have the same FRR survival rates as other epibenthic species. Therefore, for this report, the modified values for the epibenthic group (given in Section 5.7.2 were used here for shellfish (Table 24).

Table 24 Proportion mortality by species through the SZC drum and band screens

	Proportion lost		
	Drum	Band	Species group
Brown shrimp	0.206	0.206	epibenthic
Brown crab	0.206	0.206	epibenthic
Lobster	0.206	0.206	epibenthic

8.2 EAV conversions

An EAV value was calculated for brown crab. EAVs could not be calculated for brown shrimp and lobster due to a lack of size data and other biological data. For those species an EAV of 1 was used (Table 25), giving rise to impingement overestimates.

Table 25 EAV metrics and mean weight of individuals used to convert the numbers impinged to adult equivalent numbers and weights of shellfish at SZC. (See BEEMS Technical Report TR383 for brown crab EAV calculations)

Species	EAV	Mean weight per individual (kg)	Data source for mean weight
Brown shrimp	1.000	0.0013	Calculated from modelled mean number and mean weight of brown shrimp at SZB
Brown crab	0.219	0.5	Mean male and female weights at the minimum landings size, (87mm CW), and averaged assuming a 50:50 sex ratio
Lobster	1.000	0.379	Bannister et al. (1983)

8.3 Evaluating SZC impacts on shellfish

Assessments have been carried out by Cefas for lobsters and brown crabs in the southern North Sea, using slightly different assessment areas for the two species. Due to uncertainties in the data, SSB estimates are not available, and impingement losses were compared against the landings from the rectangles in the assessment regions (Table 26). None of the crustacean species is assigned to an ICES Working Group.

Table 26 Data sources used to provide information on relevant stock unit, landings and SSB

Species	Stock unit	Assessment type	Impingement effect comparator	Reference
Brown shrimp	Not defined	None	Landings, same ICES rectangles as lobster assessment	
Brown crab	Southern North Sea	Analytical assessment, but with many uncertainties	Landings from the Southern North Sea crab fishery unit (CFU) (as defined in Cefas 2017) ICES rectangles in the assessed area are shown in Table 27.	Cefas, 2017
Lobster	East Anglia	Analytical assessment, but with many uncertainties	Landings from East Anglia Lobster Fishery Unit (LFU). ICES rectangles in the assessed area are shown in Table 28.	Cefas, 2017b

Table 27 ICES Rectangles in Southern North Sea CFU

35F0	35F1	35F2	35F3	35F4	35F5	35F6	35F7 (no catch)	35F8
34F0	34F1	34F2	34F3	34F4				
	33F1	33F2	33F3	33F4				
32F0	32F1	32F2	32F3					

Table 28 ICES rectangles used in East Anglia LFU

35F0	35F1	35F2
34F0	34F1	34F2
	33F1	33F2
32F0	32F1	32F2
31F0	31F1	31F2

8.4 Predicted SZC impingement effects on shellfish without embedded mitigation measures

The predicted unmitigated SZC impingement effects for the three key crustacean species after adjusting to equivalent adults are given in Table 29. Predicted losses of brown shrimp and brown crab both exceeded the 1% threshold (3.0 % and 2.5 %, respectively). Unmitigated losses of lobsters were < 1 % and are therefore negligible.

Table 29 Annual mean SZC impingement predictions with no impingement mitigation for key shellfish species. Losses have been converted to adult equivalent (EAV) numbers and weights (t) and calculated as a % of the mean international landings (t). Species where the impingement weight > 1 % of the landings – given in bold) are shaded red

Species	Mean SZC	EAV number	EAV weight (t)	Mean landings (t)	% of landings
Brown shrimp	16,072,093	16,072,093	20.89	693	**3.01**
Brown crab	104,284	22,786	11.39	450	**2.53**
Lobster	43	43	0.02	114	**0.01**

8.5 Predicted SZC impingement effects with the LVSE intake heads fitted

Lobsters are not expected to derive any benefit from the proposed LVSE intake head design. Brown crab and brown shrimp would not be able to actively avoid the intake if they were in the water column, but any benefit from the proposed design has not yet been evaluated. Therefore, to be conservative, no benefit from the proposed intake head design has been assumed.

8.6 Predicted SZC impingement effects on shellfish with FRR systems fitted

With the inclusion of the FRR, the impingement losses of those species that exceeded 1 % in the absence of mitigation (brown shrimp and brown crab), were reduced to 0.6 % and 0.6 of landings, respectively (Table 30). The fitting of the FRR systems alone therefore reduces the impingement losses of all three crustacean species to below 1 %. For brown shrimp, losses are regarded as an overestimate, due to the use of an EAV of 1.

8.7 Conclusions on the effects of SZC on shellfish

Four shellfish species (brown crab, European lobster, brown shrimp and whelk were defined as key benthic species on the basis of their socio-economic importance (all four species) and their ecological importance (brown shrimp only). Of these, whelk were absent from the impingement dataset and there is no predicted impingement effects on the species. The predicted losses of the other three shellfish species are less than 1 % of landings. The use of landings as an impingement comparator is highly conservative as SSBs will be larger. As such the predicted SZC effects on all four key shellfish species assessed are considered negligible.

Table 30 Annual mean SZC impingement predictions with FRR mitigation fitted for shellfish species. Losses have been converted to adult equivalent (EAV) numbers and weights (t) and calculated as a % of either the mean international landings (t). Any species where the impingement weight > 1 % of the landings – given in bold) are shaded red

Species	Mean SZC	FRR mortality	EAV number	EAV weight (t)	Mean landings (t)	% of landings
Brown shrimp	16,072,093	3,310,851	3,310,851	4.30	693	**0.62**
Brown crab	104,284	22,608	4,940	2.47	450	**0.55**
Lobster	43	26	26	0.01	114	**0.01**

9 Consideration of climate change effects

Sea temperatures around the UK and Ireland have been warming at between 0.2 and 0.6 °C decade^{-1} over the past 30 years. Projected future changes in the temperature and chemistry of marine waters around the UK and Ireland are having, and will have, effects on the phenology (timing of lifecycle events), productivity and distribution of marine fish and shellfish (Heath *et al*., 2012). In a detailed study of terrestrial birds, butterflies and alpine herbs it was found that these species were undergoing northerly latitudinal change of 6.1 ± 2.4 km decade^{-1} and that there was an advancement of spring events of 2.3 d decade^{-1} (Parmesan and Yohe, 2003). Perry et al. (2005) described that distributions of both exploited and non-exploited North Sea fishes have responded to recent increases in sea temperature, with nearly two-thirds of species shifting in mean latitude or depth or both over 25 years. They found that species with shifting distributions have faster life cycles and smaller body sizes than non-shifting species and that the differential change between species could have consequences for predator-prey relationships. For species that shifted, the mean shift was 99 km northwards in 25y. Dulvy et al. (2008) found that North Sea winter bottom temperature had increased by 1·6 °C over 25 years and that during this period, the whole demersal fish assemblage deepened by ~3·6 m decade^{-1}. Simpson et al. (2011) found that most common northeast Atlantic fishes are responding significantly to warming with:

- Three times more species increasing in abundance with warming than declining.
- Local communities are being reorganized despite decadal stability in species composition.
- Species range shifts are the tip of iceberg compared to modification of local communities.

However, the effects of climate change on fish communities are hard to predict with accuracy because behaviour, genetic adaptation, habitat dependency and the impacts of fishing on species result in complex species responses (Heath et al., 2012). Petitgas *et al*., (2013) considered that the key issue for the significance of climate change impact on fishes is habitat availability and connectivity between lifecycle stages with climate driven changes in larval dispersion being a major unknown. Petitgas *et al*., (2013) considered that there was a significant risk for species with strict connectivity between spawning and nursery grounds.

The 2017 Marine Climate Change Impacts Partnership (MCCIP) review on fisheries describes the changes expected in fish and fisheries with climate change (Pinnegar *et al* 2017), and is summarised in this paragraph. There has been a trend in recent decades for warm-affinity species to increase in abundance, and cold-affinity species to decrease in abundance, with many cold-water species moving northwards. For example, there has been a decline in abundance of Atlantic cod (linked with fishing pressure and climate), and a general northwards shift. Mackerel have shown complex changes in recent years, but with a general north and westward shift linked with sea temperature. Sea bass, a warm-affinity species, expanded distribution and increased in numbers in the early 2000s, but fishing mortality then reduced numbers again. Similarly, anchovy has expanded distribution in the North Sea in the past decade. There are exceptions to this general trend, such as sole which has shifted distribution southwards and are able to remain in shallow North Sea waters all year around. Changes in plankton phenology has resulted in changes in timing of fish spawning with a shift of approximately 1.5 weeks earlier per decade in the southern North Sea since 1970s (Pinnegar *et al* 2017).

Modelling predicts that habitat suitability around the UK will increase this century for European squid, sea bass, pilchard, sprat, veined squid, John dory, anchovy, sole, plaice and whiting, and that it will decrease for saithe, hake, red mullet, haddock, halibut, mackerel and herring (Jones, 2013). Except for sole and whiting, the southerly distribution of all species is predicted to move northwards around the UK.

9.1 Changes in the Sizewell fish community

From data collected at the Sizewell A station in the 1980s, it is possible to observe changes in the Greater Sizewell Bay community in the 35+ year period up to the collection of the current SZB CIMP data. SZA impingement estimates in 1981-1982 were compared with those obtained for SZB in the current study. SZA numbers were adjusted for the different pumping capacities (SZA = 30.4 cumecs vs SZB = 51.5 cumecs; ratio ≈1.7), but not for any differences in intake head design.

For several species, the estimated numbers impinged by SZB are significantly higher than those estimated for SZA, even adjusting for pumping capacity (Table 31). For example, seabass at SZB are 194 x more abundant than the SZA estimate, and the species has increased its contribution to the total number of fish impinged from 0.02 % to 4.24 %. Similarly, twaite shad are 8.5 x more abundant in impingement catches now than in the 1980’s, and their contribution to the total has increased from 0.003 % to 0.03 %. River lamprey were not recorded in 40 sampling visits at SZA, but between 2009 and 2017, occurred in 33 % of all samples.

Conversely, the abundance of other species has declined. Greater pipefish *Syngnathus acus* were ranked as the 5th most abundant species by Turnpenny and Utting (1987) but in the current dataset, rank only 20th, and estimated numbers have decreased 10-fold (adjusted for pumping capacity). Similarly, Nilsson’s pipefish *S. rostellatus* dropped from a rank of 6th to 13th and estimated numbers decreased by a fifth.

Smelt abundance showed little or no change over the 35+ year period, contributing 0.18 % of the total numbers in both datasets. If the relative catching efficiency of the SZA and SZB intake heads for pelagic fish are considered, the smelt abundance may have increased since 1981/82 but the measured difference may also be due to natural variability.

Table 31 Annual estimated numbers of fish impinged by SZA in 1981-1982 (Turnpenny and Utting, 1987) and by SZB in 2009-2017 (BEEMS Technical Report TR339), the number at SZA raised to the SZB pumping capacity, the percentage of the total number impinged and the species’ rank

	Sizewell A 1981-1982				Sizewell B 2009-2017			
Species	**Number**	**Number (raised)**	**% of total numbers**	**Rank**	**Number**	**% of total numbers**	**Rank**	**Number SZB/SZA**
Seabass	685	1,160	0.02	28	224,719	4.24	4	193.6
European anchovy	240	407	0.01	37	28,849	0.54	8	71.0
River lamprey	0	0	0	-	2,624	0.05	27	-
Twaite shad	98	166	0.003	46	1,407	0.03	37	8.5
Smelt	6,764	11,459	0.18	14	9,320	0.18	18	0.8
Greater pipefish	66,074	111,935	1.77	5	6,485	0.12	20	0.1
Nilsson's pipefish	44,545	75,463	1.19	6	18,850	0.36	13	0.2
Flounder	22,855	38,718	0.61	8	14,912	0.28	14	0.4

9.2 Potential future changes

Some of the key observed trends in the Greater Sizewell Bay are likely to continue:

- Relative changes in species abundance with growing numbers for species that favour warmer water (in winter, in summer or both) and reducing abundance of species near to their southern latitudinal boundary.
- Effects on the phenology of some species (e.g. timing of the arrival of new recruits) and changes in migration patterns as some areas of the North Sea become more or less suitable habitat for each species and/or their prey.
- The presence of large numbers of juvenile species in the Greater Sizewell Bay is dependent upon the connectivity between spawning locations further offshore and their inshore nursery grounds. Some species have a lower tolerance to changes in winter temperatures than to summer temperatures (Dulvy et al., 2008; Perry et al., 2005) and it is possible that winter temperatures will reach a level such that some species may have to abandon fidelity to long established spawning locations which could produce a rapid reduction in the numbers of those species in the southern North Sea but not necessarily in the wider population biomass.

9.3 Effect on SZC impingement predictions

Differences in the two Sizewell impingement datasets show that the fish assemblage off Sizewell is changing due to a combination of climate change, changes in fishing pressure and other anthropogenic causes, both positive and negative (for example, improving water quality in continental rivers is attributed to increases in abundance of twaite shad in European rivers resulting in increases in abundance at Sizewell). SZC will efficiently sample the fish community at Sizewell. If a population increases in abundance then impingement numbers will increase, if a population declines in abundance then impingement numbers will reduce. In such circumstances climate change will have no effect on the predicted negligible effect of SZC impingement on the fish assemblage.

10 Effect of SZC entrapment on the Water Framework Directive (WFD) status of local water bodies

The test for the Water Framework Directive (WFD) compliance assessment is dependent on whether SZC has the potential to cause deterioration in the status of the surface water bodies (both within and between status classes) by adversely affecting biological, hydromorphological and/or physico-chemical quality elements. In principle, SZC entrapment could affect the fish biological quality element of two nearest transitional water bodies to Sizewell:

a. Blyth (S) at approximately 12 km to the north of Sizewell

b. Alde & Ore at approximately 25 km to the south of Sizewell

The United Kingdom Technical Advisory Group for WFD (WFD-UKTAG) has produced an assessment method for fish in transitional water bodies - the Transitional Fish Classification Index (TFCI). (UKTAG 2014). The method is not applicable to coastal water bodies.

The TFCI is a multimetric index composed of 10 individual components known as metrics and listed in Table 32.

Table 32 WFD Transitional Fish Classification Index metrics

Number	Metric	Community characteristic
1	Species composition	Species diversity and composition
2	Presence of indicator species	
3	Species relative abundance	Species abundance
4	Number of taxa that make up 90% of the abundance	
5	Number of estuarine resident taxa (ER)	Nursery function
6	Number of estuarine-dependent marine taxa (MS & MJ)	
7	Functional guild composition	
8	Number of benthic invertebrate feeding taxa	Trophic integrity
9	Number of piscivorous taxa	
10	Feeding guild composition.	

Each metric is assessed by comparing the observed metric values with those expected metric values under reference conditions. A set of reference conditions have been developed for different water body types and sampling gears (the latter does not include power station impingement which provides a much greater sampling efficiency than the alternative net-based sampling methods).

The TFCI is calculated as the sum of all metric scores and converted into an Ecological Quality Ratio (EQR) operating over a range from zero (a severe impact) to one (reference/minimally disturbed). The four class boundaries are:

- High/Good = 0.81
- Good/Moderate = 0.58
- Moderate/Poor = 0.4
- Poor/Bad = 0.2.

With exception of metric 3 in Table 32, all the other metrics are counts of the number of species in functional, feeding or indicator species groups found in the population samples. As described in Section 6.1.2 the fish abundance in the vicinity of Sizewell and in the two transitional water bodies will be subject to considerable in year and between year variability and also variability due to long term trends caused by climate change and changes in fishing pressure due to management action. Measurements of the TFCI will therefore be subject to variability and only developments that have a widescale, very large impact on the community would be expected to make any significant changes to the index. In terms of WFD water body status the following conclusions are pertinent:

a. Marine fish in the transitional waters of the Blyth and Alde Ore are considered to be part of the same stock units as fish at Sizewell and the SZC entrapment effects have been assessed as negligible (Table 18) and much smaller than natural variability in the size of fish populations and would, therefore, be expected to have no effect on the calculated WFD fish biological quality element (Table 32). (SZC would have no effect on freshwater fish in the water bodies).
b. No losses of indicator species are expected due to SZC entrapment
c. There are no predicted significant localised effects of SZC entrapment (Section 7.8.4) and none that would have any significant effect on the calculated WFD fish biological quality element.
d. There are no predicted changes due to SZC entrapment in the number of functional and feeding guilds in the transitional water bodies nor to the number of indicator species.

Given the above, no significant change in the EQR would be expected and certainly none that would result in a change in the WFD status of the Blyth (S) and Alde & Ore transitional water bodies due to SZC entrapment.

Stakeholders have queried whether the effects of smelt entrapment by SZC could affect the WFD status of the Alde Ore. Smelt is found in the Alde Ore and therefore will be one of the indicator species in metric 2 of the TFCI. If the effect of SZC was to totally eliminate smelt from the Alde Ore, there could be a small percentage change in the EQR (dependent upon the existing fish assemblage composition) but whether that would be sufficient to change the water body classification is not clear as we do not have access to the existing TFCI scores (if this calculation has been undertaken for the Alde Ore). The question is then could SZC eliminate smelt from the Alde Ore? For that to occur the majority of estuary population would have to migrate through the impingement risk zone for SZC (i.e. within a few metres of the proposed 4 intakes as a worst case). Given the spatial area available for smelt to migrate through (e.g. a 180-degree hemisphere extending from the Alde Ore estuary mouth) the likelihood of the Alde Ore smelt migrating via a trajectory that only took them to within a few metres of the SZC intakes is considered insignificant. In addition, smelt abundance at Sizewell, as indicated by the SZB impingement data, has no trend from 2009 to 2017 and has apparently not changed since 1981/82 (Section 9.1). Losses due to commercial fishing, which is much greater than that due to SZB, in the 35+ year period have not had any discernible effect on smelt numbers at Sizewell. The size of SZC entrapment losses would be virtually identical to those of SZB (97% of SZB) and the station would sample from the same smelt population. Based upon these considerations it is considered highly unlikely SZC would have any significant effect on the Alde Ore smelt population and certainly not enough to cause the population to collapse. Without such a collapse, the WFD status of the Alde Ore water body would not be affected.

11 Conclusions

Estimates of the number of fish, invertebrates and other individuals impinged by the current SZB station and predictions for the proposed SZC station were based upon 205 samples collected between February 2009 and December 2017. This extensive dataset provides a substantial amount of information on the abundance, seasonality and size structure of individuals impacted by the SZB station.

Ninety-one finfish species were recorded at Sizewell over the 9-year study period. Of these 24 species have been selected as being representative of the assemblage and which include species of importance commercially, ecologically and of conservation value (BEEMS Technical Report TR345), with some species occurring in multiple groups.

Where possible species were compared with defined stock units for that species and compared against internationally coordinated stock assessments. If population data could not be obtained, losses were compared against international landings, which represent only a portion of the total stock biomass. Such assessments are therefore considered highly conservative.

The predicted SZC impingement losses were compared against the negligible effect thresholds derived in Section 6. For some stocks a more precautionary approach was adopted of comparing SZC effects with 1% of a geographically limited subset of the entire stock. In particular, a highly precautionary approach was adopted for European eel whereby the Anglian RBD SSB was used as the stock reference due the uncertainties surrounding both the current eel stock status and its stock dynamics (Sections 6.1.6, 7.6.4). This was equivalent to adopting a negligible effects threshold of approximately 0.005% SSB for the eel stock. This type of precautionary assessment of SZC effects against partial stock estimates has also been applied for twaite shad, allis shad, cucumber smelt and river lamprey

For most key finfish species that will be impinged at SZC, the losses are predicted to be less than the 1 % negligible effect threshold in the absence of impingement mitigation. However, in the absence of mitigation losses of seabass, thin-lipped grey mullet and European eel were greater than 1 %. In addition, losses of brown shrimp and brown crab were greater than 1 % when compared with landings.

When the effects of the proposed LVSE intake heads and FRR systems at SZC are considered, the losses of all species are predicted to be below the 1 % negligible effects screening threshold when compared against the relevant population or landings estimates.

The individual entrainment and impingement impacts are such that when combined into a single entrapment estimate, there is very little difference to the overall conclusions that are reached when each is viewed separately. In the absence of mitigation, species that exceed the 1 % threshold are bass, thin-lipped grey mullet, European eels and sand gobies. With the proposed impingement mitigation fitted, the only species that remains above the 1 % threshold is the sand goby (entrainment = 1.4 % of abundance; impingement = 0.0 %; entrapment = 1.4 %).

Sand gobies are a highly, short-lived genus. Given that the species is not commercially-exploited, it is considered that losses of 10 % are a more appropriate negligible effects threshold (as discussed in Section 6.1). Therefore, losses of 1.4 % of total abundance by SZC are regarded as negligible.

It is therefore concluded that the proposed SZC station with FRR systems and LVSE intake heads fitted will result in negligible entrapment losses of all 24 key finfish taxa and 4 shellfish taxa.

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Appendix A Cooling water system design

The cooling water system for SZC will essentially be the same as for HPC. (There will be detailed differences for example in the layout of the FRR system, the need or not for an Archimedes screw but none of these will affect the calculations in this report). The key design features are:

a. the total cooling water abstraction at SZC will be approximately 131.86 cumecs with a maximum of 9% of the total cooling water flow supplying the essential and auxiliary cooling water systems via band screens and the remaining 91% (120 cumecs) supplying the main cooling water systems (CRF) via the station drum screens.

b. the SZC band screens will be fitted with their own FRR systems

c. the trash rack bar spacing for SZC will be 75 mm. The SZC trash rack will have a rake which returns impinged materials (including fish) to the FRR system.

d. the SZC FRR system will not be chlorinated unless there is a major change in the future water quality conditions that would facilitate the rapid growth of biofouling organisms but this is considered unlikely.

A.1 Main cooling water systems in each pumping station

SZC will consist of two EPR units. Each unit has its own forebay, pumping station, debris recovery building (HCB) and discharge pond. Each pumping station is divided into four distinct sectors: two central sectors (four channels (or ‘trains’) each) with high flow volume drum screens (ds2 and ds3) and two lateral sectors (one channel (or ‘train’) each) with lower flow volume band screens (bs1 and bs4).

Each pumping station supplies seawater to a number of systems; the main ones of which are:

CRF: Cooling Water System used to extract waste heat from the turbine steam condensers.

SEC: Essential Cooling Water system (Nuclear Island)

SEN: Auxiliary Cooling Water system (Conventional Island)

SRU: Ultimate cooling water system (Emergency use only)

CFI: Circulating Water Filtration system: supplies wash water for the drum and band screens.

The schematic layout of each pump station is shown in Figure 12.

At Mean Sea Level (MSL) the system flow rates per unit are as follows:

CRF 2*30 cumecs per unit (supplied from the 2 drum screens in each pump station)

SEC 2*1.2 cumecs per unit (can be supplied from the drum screens or band screens in any combination)

SEN 2*1.61 cumecs per unit (normally supplied from the 2 band screens in each pump station)

SRU Negligible flow (only used when testing the system or in emergency)

CFI additional to SEC flow consisting of 2*0.117 cumecs for the 2 drum screens and a worst case of 2*0.039 cumecs for the 2 band screens.

As the SEC/CFI seawater sources can be from the drum screens or band screens there is a range of different water flows through the different filtration systems at SZC.

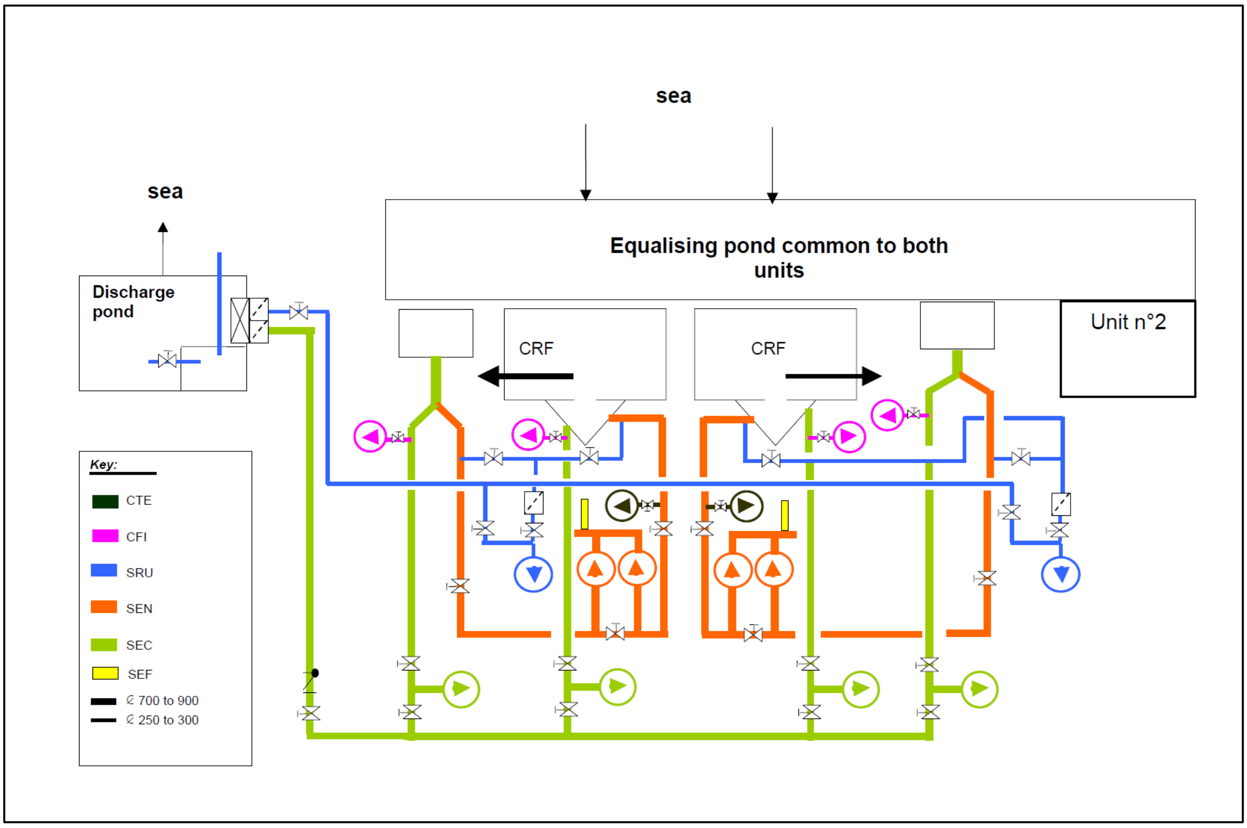


Figure 12 Illustrative schematic of EPR cooling water circuits for each unit (Source EDF CNEPE E.T.DOMA/09 0119 A1 Approved). The equalising pond shown in the figure is the station forebay and SZC has 1 forebay for each unit

Table 33 details the minimum flow at MSL (mean sea level) through the drum screens and Table 34 shows the maximum flow through the drum screens at MSL. Dependent upon the system configuration the seawater flow through the band screens can, therefore, vary between 4.9% and 9% of the total seawater abstraction of 131.86 cumecs.

Table 33 Cooling water flow volumes when SEC/CFI systems are supplied from the band screens

	Channel	flow (cumecs)	Flow through	cumecs		cumecs
	bs1	2.966	drum screens	60		
	ds2	30	band screens	5.932		
	ds3	30	Total CW flow	65.932	of which CRF	60
	bs4	2.966				
Total flow/EPR		65.932				
	2 EPRs	131.86	Flow through drum screens			120
			Total CW flow			131.86
			Band screen flow as % of total flow			**9.0%**

Table 34 Cooling water flow volumes when SEC/CFI systems are supplied from the drum screens

	Channel	flow (cumecs)	Flow through	cumecs		cumecs
	bs1	1.61	drum screens	62.712		
	ds2	31.356	band screens	3.22		
	ds3	31.356	total CW flow	65.932	of which CRF	60
	bs4	1.61				
Total flow/EPR		65.932				
	2 EPRs	131.86	Flow through drum screens			125.42
			Total CW flow			**131.86**
			Band screen flow as % of total flow			**4.9%**

Appendix B Calculated annual impingement by number at SZB and SZC without mitigation – all species

Annually raised and unmitigated number of individuals that are estimated to be impinged by SZB and predicted to be impinged by SZC, based on data from 2009-2017. Colour-coding indicates the 24 key finfish species in the Greater Sizewell Bay (BEEMS Technical Report TR345) and the type of model that was used to estimate impingement numbers for each species.

Key species
Zero-Inflated Negative Binomial with two factors for Month and Year
Zero-Inflated Negative Binomial with two factors for Sixth (2 months) and Year
Zero-Inflated Negative Binomial with two factors for Quarter and Year
Negative Binomial with one factor for Month
Species where the model did not converge

	Common name	Scientific name	SZB - estimate			SZC - prediction		
			Mean	**Lower**	**Upper**	**Mean**	**Lower**	**Upper**
	Finfish							
1	Sprat	*Sprattus sprattus*	2,782,934	1,089,329	7,110,821	7,125,393	2,789,105	18,206,464
2	Herring	*Clupea harengus*	998,201	109,735	9,081,550	2,555,783	280,965	23,252,294
3	Whiting	*Merlangius merlangus*	728,597	178,790	2,969,271	1,865,492	457,773	7,602,486
4	European seabass	*Dicentrarchus labrax*	224,719	61,132	826,468	575,367	156,523	2,116,078
5	Sand gobies	*Pomatoschistus spp*	149,045	40,410	549,860	381,612	103,464	1,407,855
6	Dover sole	*Solea solea*	97,665	39,070	244,218	250,059	100,033	625,293
7	Dab	*Limanda limanda*	58,163	8,659	390,758	148,921	22,171	1,000,493
8	European anchovy	*Engraulis encrasicolus*	28,849	4,627	180,746	73,865	11,847	462,780
9	Thin-lipped grey mullet	*Liza ramada*	26,435	578	1,209,172	67,684	1,480	3,095,949
10	Lesser weever fish	*Echiichthys (trachinus) vipera*	20,728	8,605	50,050	53,072	22,032	128,148
11	Bib	*Trisopterus luscus*	20,054	4,152	96,904	51,345	10,630	248,112
12	Transparent goby	*Aphia minuta*	19,207	2,535	156,592	49,176	6,490	400,937

	Common name	Scientific name	SZB - estimate Mean	Lower	Upper	SZC - prediction Mean	Lower	Upper
13	Nilsson's pipefish	*Syngnathus rostellatus*	18,850	5,558	64,273	48,263	14,232	164,564
14	Flounder	*Platichthys flesus*	14,912	6,298	35,319	38,180	16,125	90,431
15	Pogge (hooknose)	*Agonus cataphractus*	12,622	3,644	43,731	32,317	9,330	111,968
16	Five-bearded rockling	*Ciliata mustela*	10,586	3,600	31,175	27,103	9,218	79,820
17	European plaice	*Pleuronectes platessa*	9,877	3,306	29,540	25,288	8,464	75,633
18	Cucumber smelt	*Osmerus eperlanus*	9,320	3,216	27,101	23,863	8,233	69,389
19	Atlantic cod	*Gadus morhua*	6,579	1,678	26,056	16,845	4,297	66,715
20	Great pipefish	*Syngnathus acus*	6,485	1,644	25,599	16,605	4,209	65,544
21	Lesser spotted dogfish	*Scyliorhinus canicula*	5,184	532	50,717	13,273	1,363	129,855
22	Common sea snail	*Liparis liparis*	4,893	2,528	9,820	12,528	6,473	25,143
23	Common dragonet	*Callionymus lyra*						
24	Thornback ray	*Raja clavata*	4,219	1,185	15,492	10,802	3,034	39,665
25	Tub gurnard	*Trigla (chelidonichthys) lucerna*	3,518	1,065	12,055	9,009	2,726	30,865
26	Common sandeel	*Ammodytes tobianus*	3,175	1,366	7,547	8,128	-	-
27	River lamprey	*Lampetra fluviatilis*	2,624	929	7,935	6,720	2,378	20,316
28	Grey mullets	*Mugilidae*	2,556	2,335	2,826	6,545	5,979	7,235
29	Scald fish	*Arnoglossus laterna*	2,393	906	7,635	6,126	2,321	19,548
30	Pilchard	*Sardina pilchardus*	2,037	399	11,884	5,216	1,021	30,428
31	Starry smooth hound	*Mustelus asterias*	1,869	109	55,020	4,785	278	140,872
32	Bullrout	*Myoxocephalus scorpius*	1,824	388	9,225	4,671	994	23,619
33	Three-spined stickleback	*Gasterosteus aculeatus*	1,810	400	8,339	4,633	1,025	21,351
34	European eel	*Anguilla anguilla*	1,764	714	5,015	4,516	1,828	12,841
35	Horse-mackerel	*Trachurus trachurus*	1,592	312	11,890	4,077	800	30,442
36	Lemon sole	*Microstomus kitt*	1,563	210	12,993	4,001	538	33,267
37	Twaite shad	*Alosa fallax*	1,407	224	9,275	3,601	575	23,747
38	Poor cod	*Trisopterus minutus*	1,109	37	36,200	2,840	95	92,686
39	Sea scorpion	*Taurulus bubalis*	979	328	3,009	2,507	840	7,703
40	Tompot blenny	*Parablennius gattorugine*						
41	Brill	*Scophthalmus rhombus*	705	47	56,049	1,805	120	143,506
42	Black goby	*Gobius niger*	703	172	2,938	1,799	440	7,521
43	Solenette	*Buglossidium luteum*	646	171	3,435	1,654	437	8,796

	Common name	Scientific name	SZB - estimate			SZC - prediction		
			Mean	Lower	Upper	Mean	Lower	Upper
44	Butter fish	*Pholis gunnellus*	636	363	1,125	1,628	929	2,879
45	Snake pipefish	*Entelurus aequoreus*	634	161	2,533	1,623	412	6,485
46	Sand Smelt	*Atherina boyeri*	595	33	11,255	1,523	85	28,817
47	Great sandeel	*Hyperoplus lanceolatus*	570	44	7,434	1,459	112	19,034
48	Rock goby	*Gobius paganellus*	458	354	641	1,171	908	1,642
49	Witch	*Glyptocephalus cynoglossus*	290	5	15,990	743	14	40,941
50	Mackerel	*Scomber scombrus*	245	180	352	628	460	900
51	Red mullet	*Mullus surmuletus*	186	59	613	476	150	1,569
52	Montague's seasnail	*Liparis montagui*						
53	Garfish	*Belone belone*	80	17	396	206	42	1,015
54	Grey gurnard	*Eutrigla (chelidonicthys) gurnardus*	76	10	568	193	26	1,455
55	Turbot	*Scophthalmus maximus (Psetta maxima)*	66	10	438	169	26	1,122
56	Frie's goby	*Lesueurigobius friesii*	56	34	103	144	87	263
57	Lesser forkbeard (tadpolefish)	*Raniceps raninus*	48	23	114	122	58	291
58	Corkwing wrasse	*Crenilabrus melops*	48	22	114	122	57	292
59	John dory	*Zeus faber*	44	20	102	113	51	261
60	Sand smelt	*Atherina presbyter*	41	21	97	106	54	247
61	Northern rockling	*Ciliata septentrionalis*	37	19	78	95	48	200
62	Ballan wrasse	*Labrus bergylta*	28	11	85	71	29	217
63	Tope	*Galeorhinus galeus*	25	12	51	64	31	132
64	Four-bearded rockling	*Enchelyopus cimbrius*	25	10	73	63	24	188
65	Saithe	*Pollachius virens*	23	13	41	59	33	104
66	Lumpsucker	*Cyclopterus lumpus*	22	7	84	56	18	215
67	Spotted ray	*Raja montagui*	22	10	48	55	26	124
68	Sandeel	*Ammodytes marinus*	17	7	41	44	18	104
69	Crystal goby	*Crystallogobius linearis*	17	9	32	44	23	83
70	Thick-lipped grey mullet	*Crenimugil labrosus*	17	8	35	43	21	90
71	Norway bullhead	*Micrenophrys lilljeborgii*	15	6	44	39	16	112
72	Black seabream	*Spondyliosoma cantharus*	13	6	30	33	15	76
73	Cuckoo wrasse	*Labrus mixtus*	9	4	22	24	10	57
74	Bigeye rockling	*Antonogadus macropthalmus*	9	2	43	22	5	110

			SZB - estimate			SZC - prediction		
	Common name	Scientific name	Mean	Lower	Upper	Mean	Lower	Upper
75	Deep-snouted pipefish	*Syngnathus typhle*	7	3	18	18	8	45
76	Goldsinny	*Ctenolabrus rupestris*	6	2	16	15	6	40
77	Snake blenny	*Lumpenus lampretaeformis*	6	2	16	15	6	40
78	Norway pout	*Trisopterus esmarkii*	5	1	19	12	3	49
79	Red gurnard	*Aspitrigla (chelidonichthys) cuculus*	4	1	18	10	2	47
80	Sea Trout	*Salmo trutta*	4	1	15	10	2	37
81	Shore rockling	*Gaidropsarus mediterraneus*	3	1	27	9	2	68
82	Sand sole	*Pegusa lascaris*	2	0	13	6	1	34
83	Allis shad	*Alosa alosa*	2	0	13	5	1	33
84	Sea lamprey	*Petromyzon marinus*	2	0	13	5	1	33
85	Spotted dragonet	*Callionymus maculatus*	2	0	11	4	1	27
86	Pollack	*Pollachius pollachius*	2	0	11	4	1	27
87	Eelpout (Viviparous blenny)	*Zoarces viviparous*	‘-	‘-	‘-	‘-	‘-	‘-
88	Sandeels	*Ammodytidae*	‘-	‘-	‘-	‘-	‘-	‘-
89	Jeffrey's goby	*Buenia jeffreysii*	‘-	‘-	‘-	‘-	‘-	‘-
90	Unidentified herrings	*Clupeidae*	‘-	‘-	‘-	‘-	‘-	‘-
91	Baillons wrasse	*Symphodus (crenilabrus) balloni*	‘-	‘-	‘-	‘-	‘-	‘-
	Invertebrates							
1	Ctenophores		50,900,460	16,324,772	162,596,067	130,324,945	41,797,756	416,309,076
2	Brown shrimp	*Crangon crangon*	6,277,209	3,296,740	11,966,573	16,072,093	8,440,934	30,639,073
3	Pink shrimp	*Pandalus montagui*	1,168,116	636,333	2,148,927	2,990,831	1,629,259	5,502,088
4	Common prawn	*Palaemon serratus*	952,055	154,445	5,869,354	2,437,631	395,440	15,027,824
5	Common swimming crab	*Polybius (liocarcinus) holsatus*	438,873	81,506	2,363,663	1,123,684	208,686	6,051,895
6		*Crangon allmanni*	432,361	69,659	2,693,635	1,107,012	178,354	6,896,751
7	Plumose anemone	*Metridium senile*	218,034	106,691	450,021	558,251	273,170	1,152,229
8	Dahlia anemone	*Urticina (tealia) felina*	73,661	29,573	184,397	188,600	75,718	472,128
9	Jellyfish		69,056	10	560,031,984	176,809	26	1,433,899,368
10	Isopod	Idoteidae	62,363	8,605	463,125	159,675	22,032	1,185,779
11	Edible crab	*Cancer pagurus*	40,730	8,556	193,949	104,284	21,906	496,584
12	Little cuttlefish	*Sepiola atlantica*	32,680	22,566	47,862	83,674	57,779	122,545

	Common name	Scientific name	SZB - estimate Mean	Lower	Upper	SZC - prediction Mean	Lower	Upper
13	Ragworms	*Nereis* spp.	20,243	6,658	75,351	51,829	17,046	192,928
14	Green shore crab	*Carcinus maenas*	19,866	4,951	79,943	50,864	12,676	204,684
15	Velvet swimming crab	*Necora puber*	18,650	958	364,060	47,751	2,453	932,134
16	Unidentified spider crab	Macropodia spp	16,148	1,904	140,715	41,346	4,875	360,285
17	European common squid	*Loligo (alloteuthis) subulata*	13,960	8,071	24,849	35,744	20,664	63,623
18	Edible mussel	*Mytilus edulis*	13,933	12,967	3.59×10^{122}	35,674	33,202	9.18×10^{122}
19	Long-leg spider crab	*Macropodia rostrata*	9,999	1,406	72,015	25,602	3,600	184,387
20	Scaleworms		9,339	3,037	29,522	23,911	7,775	75,587
21	Hairy crab	*Pilumnus hirtellus*	6,649	74	603,010	17,025	189	1,543,940
22	Anemone unidentified	Anemone unidentified	5,282	4,666	6,006	13,524	11,946	15,379
23	Common starfish	*Asterias rubens*	4,776	120	189,514	12,229	308	485,231
24	Lug-worm	*Arenicola marina*	1,888	149	25,723	4,833	381	65,861
25	Sea slugs	Nudibranchia	1,209	24	67,265	3,096	61	172,223
26	Beadlet anemone	*Actinia equina*	667	501	915	1,708	1,281	2,344
27	Unidentified sea urchin		618	525	733	1,581	1,345	1,877
28	Hermit crab	*Eupagurus bernhardus*	578	485	724	1,479	1,242	1,855
29		*Processa canaliculata*	563	428	758	1,442	1,097	1,941
30		*Psammechinus miliaris*	515	357	796	1,318	914	2,039
31	Xanthidae	Xanthid crab	62	27	148	159	69	379
32	European squid	*Loligo vulgaris*	59	42	83	152	107	214
33	Northern squid	*Loligo forbesi*	45	27	78	116	70	199
34	Necklace shell	*Euspira (polinices) catena*	35	20	61	90	52	156
35	Bristle worms	Polychaeta	35	17	73	88	42	186
36		Processidae	27	14	55	69	36	141
37		*Upogebia deltaura*	21	8	55	53	20	141
38		*Ophiura ophiura*	17	6	46	44	16	117
39	Lobster	*Homarus gammarus*	17	6	58	43	15	149
40		Gammaridae	12	3	48	31	8	123
41	Purple heart urchin	*Spatangus purpureus*	11	4	30	29	11	77
42	Sea cucumbers	Holothuroidea	7	1	50	18	2	127
43	Cuttle-fish	*Sepia elegans*	5	1	37	13	2	94

			SZB - estimate			SZC - prediction		
	Common name	**Scientific name**	**Mean**	**Lower**	**Upper**	**Mean**	**Lower**	**Upper**
44		*Axius stirhynchus*	2	1	14	6	2	36
45	Unidentified swimming crab	*Liocarcinus* spp.	2	0	13	6	1	34
46	Great spider crab	*Hyas araneus*	‘-	‘-	‘-	‘-	‘-	‘-
47	Contracted crab	*Hyas coarctatus*	‘-	‘-	‘-	‘-	‘-	‘-
48	Swimming crab	*Liocarcinus depurator*	‘-	‘-	‘-	‘-	‘-	‘-
49	Marbled swimming crab	*Liocarcinus marmoreus*	‘-	‘-	‘-	‘-	‘-	‘-
50	Dwarf-swimming crab	*Liocarcinus pusillus*	‘-	‘-	‘-	‘-	‘-	‘-
51	Ghost shrimp	*Pasiphaea sivado*	‘-	‘-	‘-	‘-	‘-	‘-
52	Hairy crab	*Pilumnus spinifer*	‘-	‘-	‘-	‘-	‘-	‘-
53	Common cuttlefish	*Sepia officinalis*	‘-	‘-	‘-	‘-	‘-	‘-
54	Unidentified brittlestar		‘-	‘-	‘-	‘-	‘-	‘-
55		*Bolocera tuediae*	‘-	‘-	‘-	‘-	‘-	‘-
56	Crangonid (brown) shrimps	Crangonidae	‘-	‘-	‘-	‘-	‘-	‘-
57	Scorpion spider crab	*Inachus dorsettensis*	‘-	‘-	‘-	‘-	‘-	‘-
58	Slender spider crab	*Macropodia tenuirostris*	‘-	‘-	‘-	‘-	‘-	‘-
59	Spider crabs	Majidae	‘-	‘-	‘-	‘-	‘-	‘-
60	Opossum shrimps	Mysidacea	‘-	‘-	‘-	‘-	‘-	‘-
61	Long clawed porcelain crab	*Pisidia longgicornis*	‘-	‘-	‘-	‘-	‘-	‘-
62	Rissos crab	*Xantho pilipes*	‘-	‘-	‘-	‘-	‘-	‘-

Appendix C Mean, lower and upper numbers of fish estimated (SZB) and predicted SZC) to be impinged annually – full calculation tables

C.1 Predicted impingement without embedded mitigation measures

Species	Annually raised SZB estimate			Annually raised SZC prediction			EAV equivalent numbers		
	Mean	Lower	Upper	Mean	Lower	Upper	Mean	Lower	Upper
Sprat	2,782,934	1,089,329	7,110,821	7,125,393	2,789,105	18,206,464	5,352,978	2,095,325	13,677,673
Herring	998,201	109,735	9,081,550	2,555,783	280,965	23,252,294	1,827,944	200,951	16,630,477
Whiting	728,597	178,790	2,969,271	1,865,492	457,773	7,602,486	664,261	163,003	2,707,077
Bass	224,719	61,132	826,468	575,367	156,523	2,116,078	128,861	35,055	473,921
Sand goby	149,045	40,410	549,860	381,612	103,464	1,407,855	381,612	103,464	1,407,855
Sole	97,665	39,070	244,218	250,059	100,033	625,293	53,233	21,295	133,113
Dab	58,163	8,659	390,758	148,921	22,171	1,000,493	66,211	9,857	444,827
Anchovy	28,849	4,627	180,746	73,865	11,847	462,780	71,952	11,540	450,798
Thin-lipped grey mullet	26,435	578	1,209,172	67,684	1,480	3,095,949	5,642	123	258,069
Flounder	14,912	6,298	35,319	38,180	16,125	90,431	17,631	7,446	41,759
Plaice	9,877	3,306	29,540	25,288	8,464	75,633	8,734	2,923	26,122
Smelt	9,320	3,216	27,101	23,863	8,233	69,389	18,170	6,269	52,834
Cod	6,579	1,678	26,056	16,845	4,297	66,715	6,049	1,543	23,958
Thornback ray	4,219	1,185	15,492	10,802	3,034	39,665	2,082	585	7,646
River lamprey	2,624	929	7,935	6,720	2,378	20,316	6,720	2,378	20,316
Eel	1,764	714	5,015	4,516	1,828	12,841	4,516	1,828	12,841
Twaite shad	1,407	224	9,275	3,601	575	23,747	3,601	575	23,747
Horse mackerel	1,592	312	11,890	4,077	800	30,442	4,077	800	30,442
Mackerel	245	180	352	628	460	900	628	460	900
Tope	25	12	51	64	31	132	64	31	132
Sea trout	4	1	15	10	2	37	10	2	37
Allis shad	2	0	13	5	1	33	5	1	33
Sea lamprey	2	0	13	5	1	33	5	1	33
Salmon	0	0	0	0	0	0	0	0	0

Species	EAV equivalent weight (t)			Mean SSB (t)	EAV weight as % of SSB			Mean landings (t)	EAV weight as % of landings		
	Mean	Lower	Upper		Mean	Lower	Upper		Mean	Lower	Upper
Sprat	56.2	22.0	143.7	220,757	**0.03**	**0.01**	**0.07**	151,322	0.04	0.01	0.09
Herring	344.9	37.9	3,137.6	2,198,449	**0.02**	**0.00**	**0.14**	400,244	0.09	0.01	0.78
Whiting	189.9	46.6	773.7	151,881	**0.13**	**0.03**	**0.51**	17,570	1.08	0.27	4.40
Bass	197.3	53.7	725.5	14,897	**1.32**	**0.36**	**4.87**	3,051	6.47	1.76	23.78
Sand goby	0.7	0.2	2.7	205,882,353	**0.19**	**0.05**	**0.68**	NA	NA	NA	NA
Sole	11.4	4.6	28.5	43,770	**0.03**	**0.01**	**0.07**	12,800	0.09	0.04	0.22
Dab	2.7	0.4	18.2	NA	NA	NA	NA	6,135	**0.04**	**0.01**	**0.30**
Anchovy	1.5	0.2	9.4	NA	NA	NA	NA	1,625	**0.09**	**0.01**	**0.58**
Thin-lipped grey mullet	2.9	0.1	134.2	NA	NA	NA	NA	120	**2.45**	**0.05**	**111.90**
Flounder	1.4	0.6	3.4	NA	NA	NA	NA	2,309	**0.06**	**0.03**	**0.15**
Plaice	2.1	0.7	6.4	690,912	**0.00**	**0.00**	**0.00**	80,367	0.00	0.00	0.01
Smelt	0.3	0.1	0.9	105,733,825	**0.02**	**0.01**	**0.05**	8	3.56	1.23	10.36
Cod	15.7	4.0	62.3	103,025	**0.02**	**0.00**	**0.06**	34,701	0.05	0.01	0.18
Thornback ray	6.6	1.9	24.4	NA	NA	NA	NA	1,573	**0.42**	**0.12**	**1.55**
River lamprey	0.4	0.1	1.6	62	**0.86**	**0.30**	**2.59**	1	47.65	16.86	144.07
Eel	1.5	0.6	4.2	79	**1.89**	**0.76**	**5.37**	14	10.70	4.33	30.41
Twaite shad	1.1	0.2	7.4	7,519,986	**0.05**	**0.01**	**0.32**	1	84.60	13.50	557.82
Horse mackerel	0.6	0.1	4.3	NA	NA	NA	NA	20,798	**0.00**	**0.00**	**0.02**
Mackerel	0.2	0.1	0.3	3,888,854	**0.00**	**0.00**	**0.00**	1,026,828	0.00	0.00	0.00
Tope	0.4	0.2	0.9	NA	NA	NA	NA	498	**0.09**	**0.04**	**0.18**
Sea trout	0.0	0.0	0.1	NA	NA	NA	NA	39,795	**0.02**	**0.01**	**0.09**
Allis shad	0.0	0.0	0.0	27,397	0.02	0.00	0.12	0.2	**1.79**	**0.30**	**12.50**
Sea lamprey	0.0	0.0	0.0	NA	NA	NA	NA	NA	NA	NA	NA
Salmon	0.0	0.0	0.0	NA	NA	NA	NA	38,456	**0.00**	**0.00**	**0.00**

C.2 Predicted SZC impingement with the effect of LVSE intake heads

	Annually raised SZB estimate			Annually raised SZC estimate			Intake head mortality			EAV equivalent numbers		
Species	**Mean**	**Lower**	**Upper**	**Mean**	**Lower**	**Upper**	**Mean**	**Lower**	**Upper**	**Mean**	**Lower**	**Upper**
Sprat	2,782,934	1,089,329	7,110,821	7,125,393	2,789,105	18,206,464	2,729,025	1,068,227	6,973,076	2,050,190	802,510	5,238,549
Herring	998,201	109,735	9,081,550	2,555,783	280,965	23,252,294	978,865	107,610	8,905,629	700,103	76,964	6,369,473
Whiting	728,597	178,790	2,969,271	1,865,492	457,773	7,602,486	714,484	175,327	2,911,752	254,412	62,430	1,036,811
Bass	224,719	61,132	826,468	575,367	156,523	2,116,078	220,366	59,948	810,458	49,354	13,426	181,512
Sand goby	149,045	40,410	549,860	381,612	103,464	1,407,855	146,157	39,627	539,208	146,157	39,627	539,208
Sole	97,665	39,070	244,218	250,059	100,033	625,293	95,773	38,313	239,487	20,388	8,156	50,982
Dab	58,163	8,659	390,758	148,921	22,171	1,000,493	57,037	8,491	383,189	25,359	3,775	170,369
Anchovy	28,849	4,627	180,746	73,865	11,847	462,780	28,290	4,537	177,245	27,558	4,420	172,656
Thin-lipped grey mullet	26,435	578	1,209,172	67,684	1,480	3,095,949	25,923	567	1,185,748	2,161	47	98,840
Flounder	14,912	6,298	35,319	38,180	16,125	90,431	14,623	6,176	34,635	6,753	2,852	15,994
Plaice	9,877	3,306	29,540	25,288	8,464	75,633	9,685	3,242	28,967	3,345	1,120	10,005
Smelt	9,320	3,216	27,101	23,863	8,233	69,389	9,139	3,153	26,576	6,959	2,401	20,236
Cod	6,579	1,678	26,056	16,845	4,297	66,715	6,451	1,646	25,552	2,317	591	9,176
Thornback ray	4,219	1,185	15,492	10,802	3,034	39,665	4,137	1,162	15,192	797	224	2,928
River lamprey	2,624	929	7,935	6,720	2,378	20,316	2,574	911	7,781	2,574	911	7,781
Eel	1,764	714	5,015	4,516	1,828	12,841	1,730	700	4,918	1,730	700	4,918
Twaite shad	1,407	224	9,275	3,601	575	23,747	1,379	220	9,095	1,379	220	9,095
Horse mackerel	1,592	312	11,890	4,077	800	30,442	1,561	306	11,659	1,561	306	11,659
Mackerel	245	180	352	628	460	900	241	176	345	241	176	345
Tope	25	12	51	64	31	132	24	12	50	24	12	50
Sea trout	4	1	15	10	2	37	4	1	14	4	1	14
Allis shad	2	0	13	5	1	33	2	0	13	2	0	13
Sea lamprey	2	0	13	5	1	33	2	0	13	2	0	13
Salmon	0	0	0	0	0	0	0	0	0	0	0	0

	EAV equivalent weight (t)				EAV weight as % of SSB			Mean	EAV weight as % of landings		
Species	**Mean**	**Lower**	**Upper**	**Mean SSB (t)**	**Mean**	**Lower**	**Upper**	**landings (t)**	**Mean**	**Lower**	**Upper**
Sprat	21.5	8.4	55.0	220,757	**0.01**	**0.00**	**0.02**	151,322	0.01	0.01	0.04
Herring	132.1	14.5	1,201.7	2,198,449	**0.01**	**0.00**	**0.05**	400,244	0.03	0.00	0.30
Whiting	72.7	17.8	296.3	151,881	**0.05**	**0.01**	**0.20**	17,570	0.41	0.10	1.69
Bass	75.6	20.6	277.9	14,897	**0.51**	**0.14**	**1.87**	3,051	2.48	0.67	9.11
Sand goby	0.3	0.1	1.0	205,882,353	**0.07**	**0.02**	**0.26**	NA	NA	NA	NA
Sole	4.4	1.7	10.9	43,770	**0.01**	**0.00**	**0.02**	12,800	0.03	0.01	0.09
Dab	1.0	0.2	7.0	NA	NA	NA	NA	6,135	**0.02**	**0.00**	**0.11**
Anchovy	0.6	0.1	3.6	NA	NA	NA	NA	1,625	**0.04**	**0.01**	**0.22**
Thin-lipped grey mullet	1.1	0.0	51.4	NA	NA	NA	NA	120	**0.94**	**0.02**	**42.86**
Flounder	0.6	0.2	1.3	NA	NA	NA	NA	2,309	**0.02**	**0.01**	**0.06**
Plaice	0.8	0.3	2.5	690,912	**0.00**	**0.00**	**0.00**	80,367	0.00	0.00	0.00
Smelt	0.1	0.0	0.3	105,733,825	**0.01**	**0.00**	**0.02**	8	1.36	0.47	3.97
Cod	6.0	1.5	23.9	103,025	**0.01**	**0.00**	**0.02**	34,701	0.02	0.00	0.07
Thornback ray	2.5	0.7	9.4	NA	NA	NA	NA	1,573	**0.16**	**0.05**	**0.59**
River lamprey	0.2	0.1	0.6	62	**0.33**	**0.12**	**0.99**	1	18.25	6.46	55.18
Eel	0.6	0.2	1.6	79	**0.72**	**0.29**	**2.06**	14	4.10	1.66	11.65
Twaite shad	0.4	0.1	2.8	7,519,986	**0.02**	**0.00**	**0.12**	1	32.40	5.17	213.64
Horse mackerel	0.2	0.0	1.6	NA	NA	NA	NA	20,798	**0.00**	**0.00**	**0.01**
Mackerel	0.1	0.1	0.1	3,888,854	**0.00**	**0.00**	**0.00**	1,026,828	0.00	0.00	0.00
Tope	0.2	0.1	0.3	NA	NA	NA	NA	498	**0.03**	**0.02**	**0.07**
Sea trout	0.0	0.0	0.0	NA	NA	NA	NA	39,795	**0.01**	**0.00**	**0.04**
Allis shad	0.0	0.0	0.0	27,397	**0.01**	**0.00**	**0.05**	0.2	0.68	0.11	4.79
Sea lamprey	0.0	0.0	0.0	NA	NA	NA	NA	NA	NA	NA	NA
Salmon	0.0	0.0	0.0	NA	NA	NA	NA	38,456	**0.00**	**0.00**	**0.00**

C.3 Predicted SZC impingement with FRR systems fitted

	Annually raised SZB estimate			Annually raised SZC estimate			FRR mortality			EAV equivalent numbers		
Species	**Mean**	**Lower**	**Upper**	**Mean**	**Lower**	**Upper**	**Mean**	**Lower**	**Upper**	**Mean**	**Lower**	**Upper**
Sprat	2,782,934	1,089,329	7,110,821	7,125,393	2,789,105	18,206,464	7,125,393	2,789,105	18,206,464	5,352,978	2,095,325	13,677,673
Herring	998,201	109,735	9,081,550	2,555,783	280,965	23,252,294	2,555,783	280,965	23,252,294	1,827,944	200,951	16,630,477
Whiting	728,597	178,790	2,969,271	1,865,492	457,773	7,602,486	1,026,879	251,986	4,184,864	365,649	89,727	1,490,138
Bass	224,719	61,132	826,468	575,367	156,523	2,116,078	317,979	86,503	1,169,457	71,215	19,373	261,914
Sand goby	149,045	40,410	549,860	381,612	103,464	1,407,855	78,612	21,314	290,018	78,612	21,314	290,018
Sole	97,665	39,070	244,218	250,059	100,033	625,293	53,392	21,359	133,510	11,366	4,547	28,422
Dab	58,163	8,659	390,758	148,921	22,171	1,000,493	33,622	5,005	225,880	14,948	2,225	100,428
Anchovy	28,849	4,627	180,746	73,865	11,847	462,780	73,865	11,847	462,780	71,952	11,540	450,798
Thin-lipped grey mullet	26,435	578	1,209,172	67,684	1,480	3,095,949	37,266	815	1,704,591	3,106	68	142,090
Flounder	14,912	6,298	35,319	38,180	16,125	90,431	14,440	6,099	34,202	6,668	2,816	15,794
Plaice	9,877	3,306	29,540	25,288	8,464	75,633	5,680	1,901	16,989	1,962	657	5,868
Smelt	9,320	3,216	27,101	23,863	8,233	69,389	23,863	8,233	69,389	18,170	6,269	52,834
Cod	6,579	1,678	26,056	16,845	4,297	66,715	10,607	2,706	42,010	3,809	972	15,086
Thornback ray	4,219	1,185	15,492	10,802	3,034	39,665	2,277	639	8,361	439	123	1,612
River lamprey	2,624	929	7,935	6,720	2,378	20,316	1,384	490	4,185	1,384	490	4,185
Eel	1,764	714	5,015	4,516	1,828	12,841	982	398	2,793	982	398	2,793
Twaite shad	1,407	224	9,275	3,601	575	23,747	3,601	575	23,747	3,601	575	23,747
Horse mackerel	1,592	312	11,890	4,077	800	30,442	4,077	800	30,442	4,077	800	30,442
Mackerel	245	180	352	628	460	900	628	460	900	628	460	900
Tope	25	12	51	64	31	132	39	19	79	39	19	79
Sea trout	4	1	15	10	2	37	10	2	37	10	2	37
Allis shad	2	0	13	5	1	33	5	1	33	5	1	33
Sea lamprey	2	0	13	5	1	33	1	0	7	1	0	7
Salmon	0	0	0	0	0	0	0	0	0	0	0	0

	EAV equivalent weight (t)			mean SSB (t)	EAV weight as % of SSB			Mean landings (t)	EAV weight as % of landings		
Species	**Mean**	**Lower**	**Upper**		**Mean**	**Lower**	**Upper**		**Mean**	**Lower**	**Upper**
Sprat	56.2	22.0	143.7	220,757	**0.03**	**0.01**	**0.07**	151,322	0.04	0.01	0.09

	EAV equivalent weight (t)			mean SSB	EAV weight as % of SSB			Mean	EAV weight as % of landings		
Species	**Mean**	**Lower**	**Upper**	**(t)**	**Mean**	**Lower**	**Upper**	**landings (t)**	**Mean**	**Lower**	**Upper**
Herring	344.9	37.9	3,137.6	2,198,449	**0.02**	**0.00**	**0.14**	400,244	0.09	0.01	0.78
Whiting	104.5	25.6	425.9	151,881	**0.07**	**0.02**	**0.28**	17,570	0.59	0.15	2.42
Bass	109.0	29.7	400.9	14,897	**0.73**	**0.20**	**2.69**	3,051	3.57	0.97	13.14
Sand goby	0.1	0.0	0.6	205,882,353	**0.04**	**0.01**	**0.14**	NA	NA	NA	NA
Sole	2.4	1.0	6.1	43,770	**0.01**	**0.00**	**0.01**	12,800	0.02	0.01	0.05
Dab	0.6	0.1	4.1	NA	NA	NA	NA	6,135	**0.02**	**0.00**	**0.16**
Anchovy	1.5	0.2	9.4	NA	NA	NA	NA	1,625	**0.09**	**0.01**	**0.58**
Thin-lipped grey mullet	1.6	0.0	73.9	NA	NA	NA	NA	120	**1.35**	**0.03**	**61.61**
Flounder	0.5	0.2	1.3	NA	NA	NA	NA	2,309	**0.02**	**0.01**	**0.06**
Plaice	0.5	0.2	1.4	690,912	**0.00**	**0.00**	**0.00**	80,367	0.00	0.00	0.00
Smelt	0.3	0.1	0.9	105,733,825	**0.02**	**0.01**	**0.05**	8	3.56	1.23	10.36
Cod	9.9	2.5	39.3	103,025	**0.01**	**0.00**	**0.04**	34,701	0.03	0.01	0.11
Thornback ray	1.4	0.4	5.1	NA	NA	NA	NA	1,573	**0.09**	**0.03**	**0.33**
River lamprey	0.1	0.0	0.3	62	**0.18**	**0.06**	**0.53**	1	9.82	3.47	29.68
Eel	0.3	0.1	0.9	79	**0.39**	**0.16**	**1.11**	14	2.20	0.89	6.27
Twaite shad	1.1	0.2	7.4	7,519,986	**0.05**	**0.01**	**0.32**	1	84.60	13.50	557.82
Horse mackerel	0.6	0.1	4.3	NA	NA	NA	NA	20,798	**0.00**	**0.00**	**0.02**
Mackerel	0.2	0.1	0.3	3,888,854	**0.00**	**0.00**	**0.00**	1,026,828	0.00	0.00	0.00
Tope	0.3	0.1	0.5	NA	NA	NA	NA	498	**0.05**	**0.03**	**0.11**
Sea trout	0.0	0.0	0.1	NA	NA	NA	NA	39,795	**0.02**	**0.01**	**0.09**
Allis shad	0.0	0.0	0.0	27,397	**0.02**	**0.00**	**0.12**	0.2	1.79	0.30	12.50
Sea lamprey	0.0	0.0	0.0	NA	NA	NA	NA	NA	NA	NA	NA
Salmon	0.0	0.0	0.0	NA	NA	NA	NA	38,456	**0.00**	**0.00**	**0.00**

C.4 Predicted SZC impingement with the effect of LVSE intake heads and FRR systems fitted

Species	Annually raised SZB estimate			Annually raised SZC estimate			Intake head mortality		
	Mean	Lower	Upper	Mean	Lower	Upper	Mean	Lower	Upper
Sprat	2,782,934	1,089,329	7,110,821	7,125,393	2,789,105	18,206,464	2,729,025	1,068,227	6,973,076
Herring	998,201	109,735	9,081,550	2,555,783	280,965	23,252,294	978,865	107,610	8,905,629
Whiting	728,597	178,790	2,969,271	1,865,492	457,773	7,602,486	714,484	175,327	2,911,752
Bass	224,719	61,132	826,468	575,367	156,523	2,116,078	220,366	59,948	810,458
Sand goby	149,045	40,410	549,860	381,612	103,464	1,407,855	146,157	39,627	539,208
Sole	97,665	39,070	244,218	250,059	100,033	625,293	95,773	38,313	239,487
Dab	58,163	8,659	390,758	148,921	22,171	1,000,493	57,037	8,491	383,189
Anchovy	28,849	4,627	180,746	73,865	11,847	462,780	28,290	4,537	177,245
Thin-lipped grey mullet	26,435	578	1,209,172	67,684	1,480	3,095,949	25,923	567	1,185,748
Flounder	14,912	6,298	35,319	38,180	16,125	90,431	14,623	6,176	34,635
Plaice	9,877	3,306	29,540	25,288	8,464	75,633	9,685	3,242	28,967
Smelt	9,320	3,216	27,101	23,863	8,233	69,389	9,139	3,153	26,576
Cod	6,579	1,678	26,056	16,845	4,297	66,715	6,451	1,646	25,552
Thornback ray	4,219	1,185	15,492	10,802	3,034	39,665	4,137	1,162	15,192
River lamprey	2,624	929	7,935	6,720	2,378	20,316	2,574	911	7,781
Eel	1,764	714	5,015	4,516	1,828	12,841	1,730	700	4,918
Twaite shad	1,407	224	9,275	3,601	575	23,747	1,379	220	9,095
Horse mackerel	1,592	312	11,890	4,077	800	30,442	1,561	306	11,659
Mackerel	245	180	352	628	460	900	241	176	345
Tope	25	12	51	64	31	132	24	12	50
Sea trout	4	1	15	10	2	37	4	1	14
Allis shad	2	0	13	5	1	33	2	0	13
Sea lamprey	2	0	13	5	1	33	2	0	13
Salmon	0	0	0	0	0	0	0	0	0

Species	Trash rack mortality			Drum screen mortality			Band screen mortality		
	Mean	Lower	Upper	Mean	Lower	Upper	Mean	Lower	Upper
Sprat	0	0	0	2,483,413	972,087	6,345,499	245,612	96,140	627,577
Herring	700,081	76,962	6,369,275	253,694	27,889	2,308,082	25,091	2,758	228,272
Whiting	0	0	0	328,991	80,731	1,340,745	64,304	15,779	262,058
Bass	52	14	191	101,446	27,597	373,095	19,828	5,394	72,924
Sand goby	0	0	0	27,399	7,428	101,080	2,710	735	9,997
Sole	0	0	0	17,954	7,182	44,894	1,776	710	4,440
Dab	23,886	3,556	160,473	6,214	925	41,750	615	92	4,129
Anchovy	0	0	0	25,744	4,129	161,293	2,546	408	15,952
Thin-lipped grey mullet	7	0	336	11,933	261	545,835	2,332	51	106,687
Flounder	459	194	1,086	2,655	1,121	6,289	263	111	622
Plaice	0	0	0	1,816	608	5,430	180	60	537
Smelt	0	0	0	8,317	2,870	24,184	823	284	2,392
Cod	40	10	160	3,267	834	12,940	577	147	2,285
Thornback ray	0	0	0	776	218	2,848	77	22	282
River lamprey	0	0	0	482	171	1,459	48	17	144
Eel	0	0	0	324	131	922	32	13	91
Twaite shad	1,218	194	8,030	147	23	969	15	2	96
Horse mackerel	0	0	0	1,421	279	10,610	141	28	1,049
Mackerel	0	0	0	219	160	314	22	16	31
Tope	0	0	0	5	2	9	0	0	1
Sea trout	4	1	14	0	0	0	0	0	0
Allis shad	2	0	13	0	0	0	0	0	0
Sea lamprey	0	0	0	0	0	2	0	0	0
Salmon	0	0	0	0	0	0	0	0	0

Species	Total FRR mortality			EAV equivalent numbers			EAV equivalent weight (t)		
	Mean	Lower	Upper	Mean	Lower	Upper	Mean	Lower	Upper
Sprat	2,729,025	1,068,227	6,973,076	2,050,190	802,510	5,238,549	21.53	8.43	55.02

Herring	978,865	107,610	8,905,629	700,103	76,964	6,369,473	132.08	14.52	1,201.70
Whiting	393,295	96,511	1,602,803	140,044	34,365	570,723	40.03	9.82	163.12
Bass	121,326	33,005	446,211	27,172	7,392	99,934	41.60	11.32	152.98
Sand goby	30,108	8,163	111,077	30,108	8,163	111,077	0.06	0.02	0.21
Sole	19,729	7,892	49,334	4,200	1,680	10,502	0.90	0.36	2.25
Dab	30,715	4,573	206,352	13,656	2,033	91,746	0.56	0.08	3.74
Anchovy	28,290	4,537	177,245	27,558	4,420	172,656	0.57	0.09	3.58
Thin-lipped grey mullet	14,273	312	652,858	1,190	26	54,420	0.62	0.01	28.30
Flounder	3,377	1,426	7,997	1,559	659	3,693	0.13	0.05	0.30
Plaice	1,995	668	5,967	689	231	2,061	0.17	0.06	0.51
Smelt	9,139	3,153	26,576	6,959	2,401	20,236	0.12	0.04	0.33
Cod	3,884	991	15,385	1,395	356	5,525	3.63	0.93	14.38
Thornback ray	852	239	3,130	164	46	603	0.52	0.15	1.93
River lamprey	530	188	1,603	530	188	1,603	0.04	0.01	0.13
Eel	356	144	1,013	356	144	1,013	0.12	0.05	0.33
Twaite shad	1,379	220	9,095	1,379	220	9,095	0.43	0.07	2.85
Horse mackerel	1,561	306	11,659	1,561	306	11,659	0.22	0.04	1.64
Mackerel	241	176	345	241	176	345	0.08	0.06	0.11
Tope	5	2	10	5	2	10	0.03	0.02	0.07
Sea trout	4	1	14	4	1	14	0.01	0.00	0.02
Allis shad	2	0	13	2	0	13	0.00	0.00	0.01
Sea lamprey	0	0	3	0	0	3	0.00	0.00	0.00
Salmon	0	0	0	0	0	0	0.00	0.00	0.00

Note: Total FRR mortality is the sum of the trash rack, drum screen and band screen mortalities, applied to the numbers impinged after the effect of the intake head has been considered.

	mean SSB (t)	**EAV weight as % of SSB**			**Mean landings (t)**	**EAV weight as % of landings**		
Species		**Mean**	**Lower**	**Upper**		**Mean**	**Lower**	**Upper**

Sprat	220,757	**0.01**	**0.00**	**0.02**	151,322	0.01	0.01	0.04
Herring	2,198,449	**0.01**	**0.00**	**0.05**	400,244	0.03	0.00	0.30
Whiting	151,881	**0.03**	**0.01**	**0.11**	17,570	0.23	0.06	0.93
Bass	14,897	**0.28**	**0.08**	**1.03**	3,051	1.36	0.37	5.01
Sand goby	205,882,353	**0.01**	**0.00**	**0.05**	NA	NA	NA	NA
Sole	43,770	**0.00**	**0.00**	**0.01**	12,800	0.01	0.00	0.02
Dab	NA	NA	NA	NA	6,135	**0.01**	**0.00**	**0.06**
Anchovy	NA	NA	NA	NA	1,625	**0.04**	**0.01**	**0.22**
Thin-lipped grey mullet	NA	NA	NA	NA	120	**0.52**	**0.01**	**23.60**
Flounder	NA	NA	NA	NA	2,309	**0.01**	**0.00**	**0.01**
Plaice	690,912	**0.00**	**0.00**	**0.00**	80,367	0.00	0.00	0.00
Smelt	105,733,825	**0.01**	**0.00**	**0.02**	8	1.36	0.47	3.97
Cod	103,025	**0.00**	**0.00**	**0.01**	34,701	0.01	0.00	0.04
Thornback ray	NA	NA	NA	NA	1,573	**0.03**	**0.01**	**0.12**
River lamprey	62	**0.07**	**0.02**	**0.20**	1	3.76	1.33	11.37
Eel	79	**0.15**	**0.06**	**0.42**	14	0.84	0.34	2.40
Twaite shad	7,519,986	**0.02**	**0.00**	**0.12**	1	32.40	5.17	213.64
Horse mackerel	NA	NA	NA	NA	20,798	**0.00**	**0.00**	**0.01**
Mackerel	3,888,854	**0.00**	**0.00**	**0.00**	1,026,828	0.00	0.00	0.00
Tope	NA	NA	NA	NA	498	**0.01**	**0.00**	**0.01**
Sea trout	NA	NA	NA	NA	39,795	**0.01**	**0.00**	**0.04**
Allis shad	27,397	**0.01**	**0.00**	**0.05**	0	0.68	0.11	4.79
Sea lamprey	NA	NA	NA	NA	NA	NA	NA	NA
Salmon	NA	NA	NA	NA	38,456	**0.00**	**0.00**	**0.00**

Appendix D Mean numbers of fish estimated impinged annually at SZB

Annually raised mean SZB estimates of impingement for 24 key species, with no mitigation and with the embedded FRR mitigation. Total losses have been converted to adult equivalent (EAV) numbers and weights (t) and calculated as a percentage of either the mean stock SSB (t) or mean international landings (t). Species where the impingement weight exceed 1 % of the relevant stock comparator are shaded in red. Note: numbers in red font are either estimates of the population numbers (e.g. sand goby) or reported catch numbers (salmon and sea trout).

D.1 Unmitigated impingement effects

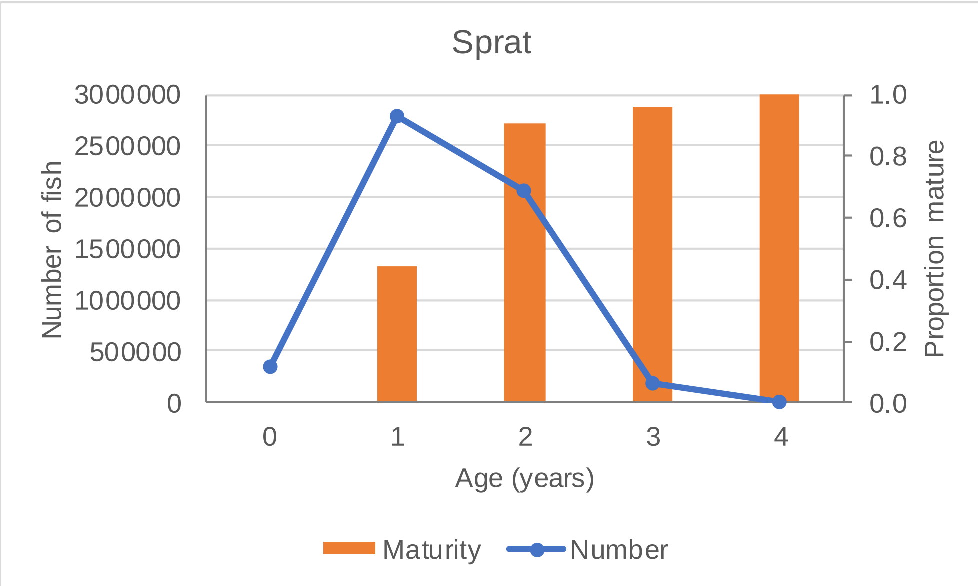
Species	Mean SZB estimate	EAV number	EAV weight (t)	Mean SSB	% of SSB	Mean landings (t)	% of landings
Sprat	2,782,934	2,090,690	21.96	220,757	**0.01**	151,322	0.01
Herring	998,201	713,932	134.69	2,198,449	**0.01**	400,244	0.03
Whiting	728,597	259,437	74.15	151,881	**0.05**	17,570	0.42
Bass	224,719	50,329	77.04	14,897	**0.52**	3,051	2.53
Sand goby	149,045	149,045	0.28	205,882,353	**0.07**	NA	NA
Sole	97,665	20,791	4.45	43,770	**0.01**	12,800	0.03
Dab	58,163	25,860	1.06	NA	NA	6,135	**0.02**
Anchovy	28,849	28,102	0.58	NA	NA	1,625	**0.04**
Thin-lipped grey mullet	26,435	2,204	1.15	NA	NA	120	**0.96**
Flounder	14,912	6,886	0.56	NA	NA	2,309	**0.02**
Plaice	9,877	3,411	0.84	690,912	**0.00**	80,367	0.00
Smelt	9,320	7,096	0.12	105,733,825	**0.01**	8	1.39
Cod	6,579	2,363	6.15	103,025	**0.01**	34,701	0.02
Thornback ray	4,219	813	2.60	NA	NA	1,573	**0.17**
River lamprey	2,624	2,624	0.21	62	**0.34**	1	18.61
Eel	1,764	1,764	0.58	79	**0.74**	14	4.18
Twaite shad	1,407	1,407	0.44	7,519,986	**0.02**	1	**33.04**
Horse mackerel	1,592	1,592	0.22	NA	NA	20,798	**0.00**
Mackerel	245	245	0.08	3,888,854	**0.00**	1,026,828	0.00
Tope	25	25	0.17	NA	NA	498	**0.03**
Sea trout	4	4	0.01	NA	NA	39,795	**0.01**
Allis shad	2	2	0.00	27,397	**0.01**	0	**0.70**
Sea lamprey	2	2	0.00	NA	NA	NA	NA
Salmon	0	0	0.00	NA	NA	38,456	**0.00**

D.2 Impingement effects with embedded FRR mitigation

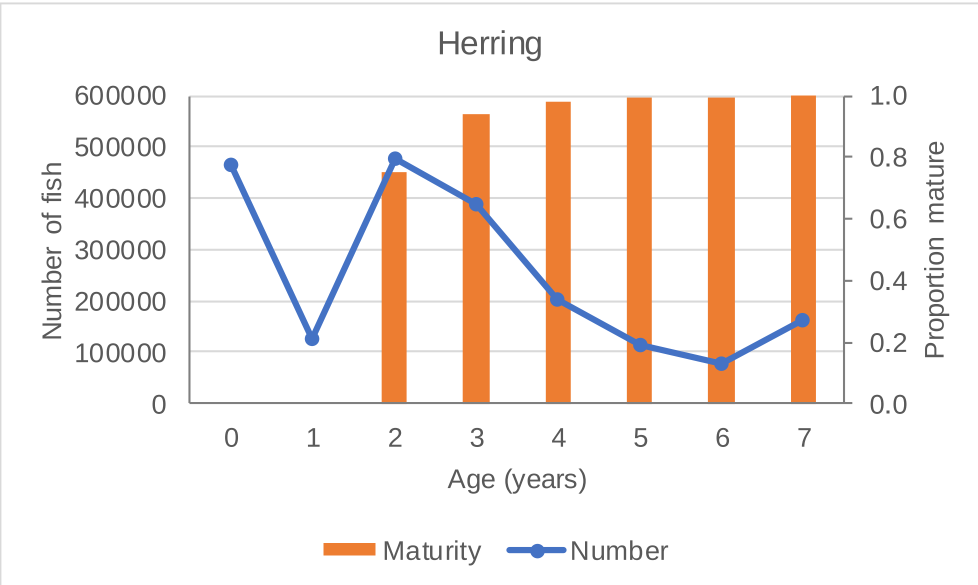
Species	Mean SZB estimate	FRR mortality	EAV number	EAV weight (t)	Mean SSB	% of SSB	Mean landings (t)	% of landings
Sprat	2,782,934	2,782,934	2,090,690	21.96	220,757	**0.01**	151,322	0.01
Herring	998,201	998,201	713,932	134.69	2,198,449	**0.01**	400,244	0.03
Whiting	728,597	364,299	129,719	37.08	151,881	**0.02**	17,570	0.21
Bass	224,719	112,359	25,164	38.52	14,897	**0.26**	3,051	1.26
Sand goby	149,045	29,809	29,809	0.06	205,882,353	**0.01**	NA	NA
Sole	97,665	19,533	4,158	0.89	43,770	**0.00**	12,800	0.01
Dab	58,163	11,633	5,172	0.21	NA	NA	6,135	**0.00**
Anchovy	28,849	28,849	28,102	0.58	NA	NA	1,625	**0.04**
Thin-lipped grey mullet	26,435	13,218	1,102	0.57	NA	NA	120	**0.48**
Flounder	14,912	2,982	1,377	0.11	NA	NA	2,309	**0.00**
Plaice	9,877	1,975	682	0.17	690,912	**0.00**	80,367	0.00
Smelt	9,320	9,320	7,096	0.12	105,733,825	**0.01**	8	1.39
Cod	6,579	3,289	1,181	3.07	103,025	**0.00**	34,701	0.01
Thornback ray	4,219	844	163	0.52	NA	NA	1,573	**0.03**
River lamprey	2,624	525	525	0.04	62	**0.07**	1	3.72
Eel	1,764	353	353	0.12	79	**0.15**	14	0.84
Twaite shad	1,407	1,407	1,407	0.44	7,519,986	**0.02**	1	**33.04**
Horse mackerel	1,592	1,592	1,592	0.22	NA	NA	20,798	**0.00**
Mackerel	245	245	245	0.08	3,888,854	**0.00**	1,026,828	0.00
Tope	25	5	5	0.03	NA	NA	498	**0.01**
Sea trout	4	2	2	0.00	NA	NA	39,795	**0.00**
Allis shad	2	2	2	0.00	27,397	**0.01**	0	**0.70**
Sea lamprey	2	0	0	0.00	NA	NA	NA	NA
Salmon	0	0	0	0.00	NA	NA	38,456	**0.00**

Appendix E Number at age and proportion maturity of commonly impinged fish species

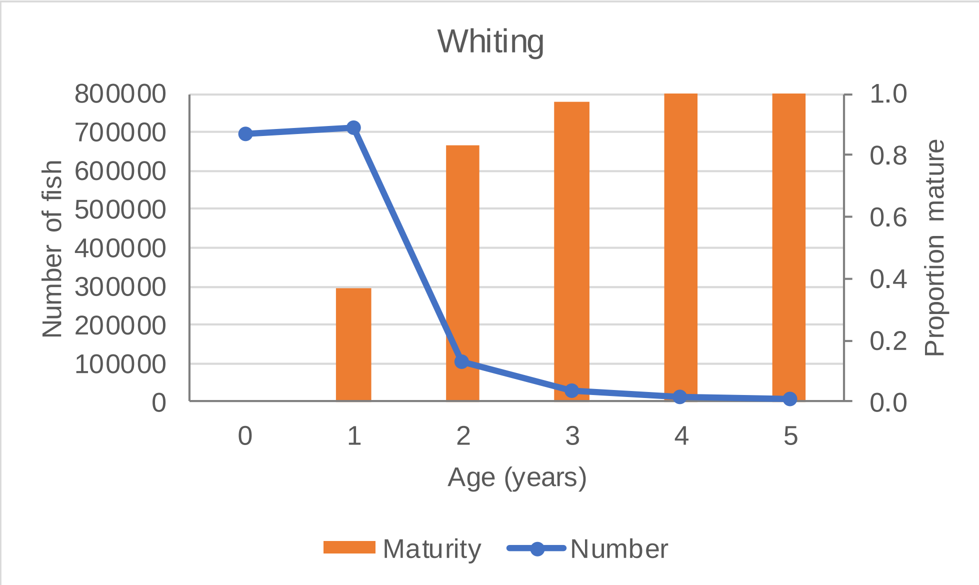
E.1 Sprat



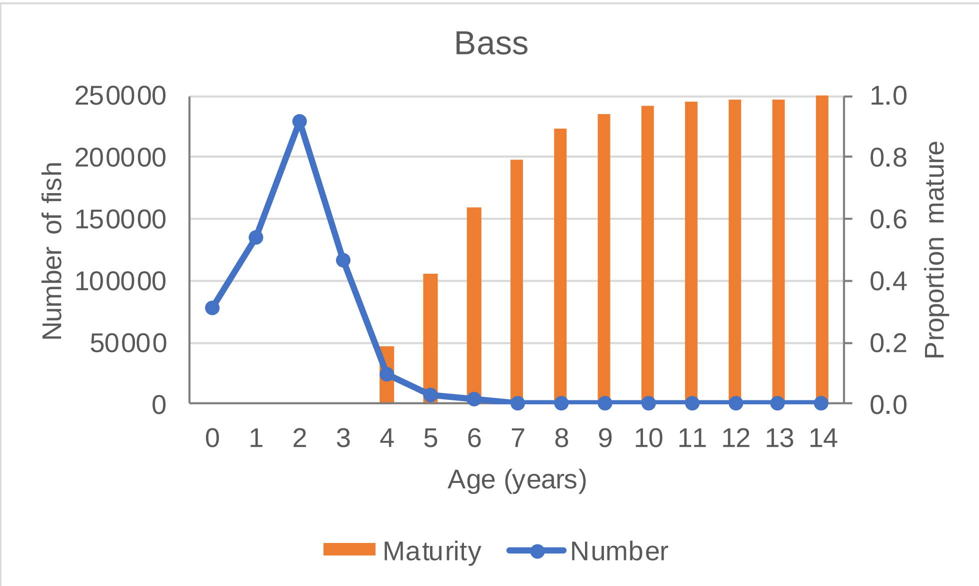
E.2 Herring



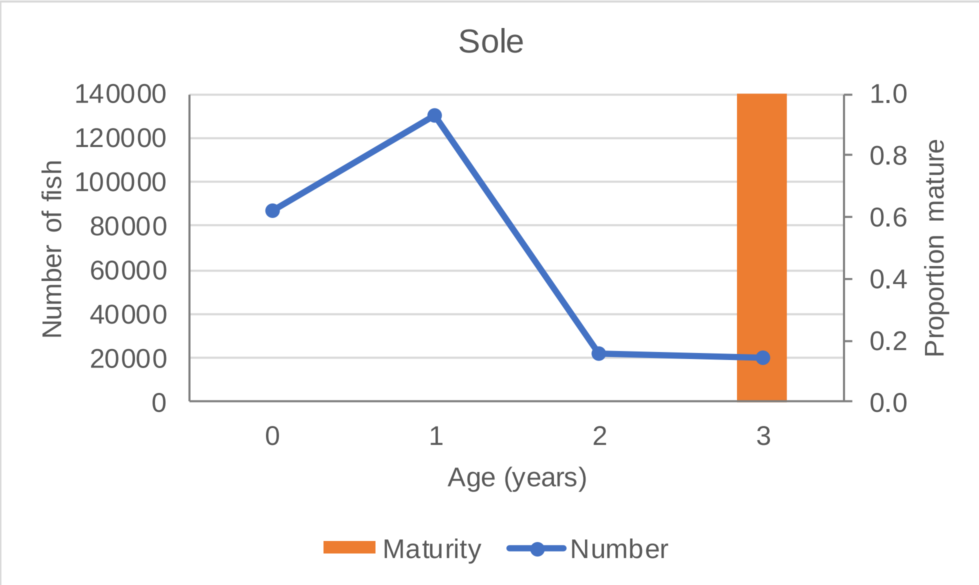
E.3 Whiting

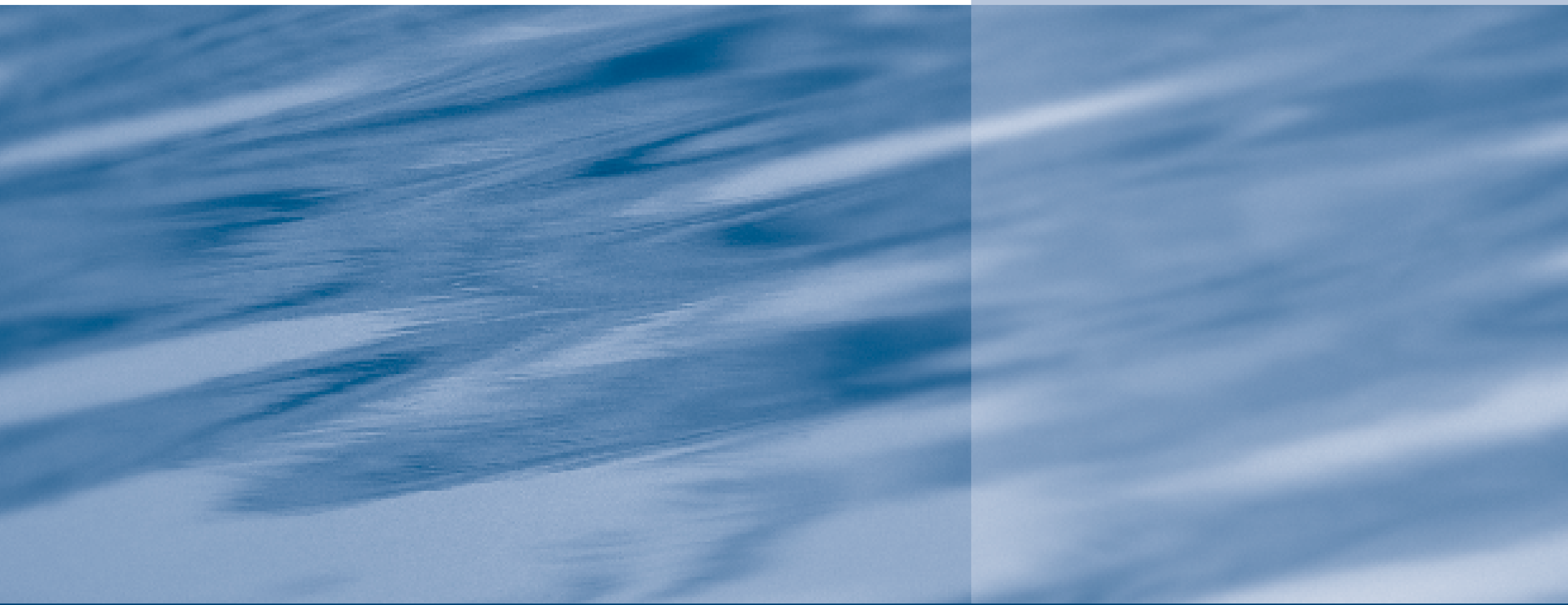


E.4 Bass



E.5 Sole


Sole
140000
120000
100000
80000
60000
40000
20000
0
Number of fish
1.0
0.8
0.6
0.4
0.2
0.0
Proportion mature
0
1
2
3
Age (years)
Maturity
Number



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Screening for Intake and Outfalls: a best practice guide

Science Report SC030231

ENGLISH NATURE

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Professor Mike Depledge Head of Science

SUMMARY

The aim of this Guide is to provide a description of the legal responsibilities of operators of water intakes and outfalls and, from a review of current, worldwide examples, to present a synopsis of methods that are known to work best for different species and lifestages of fish in different situations. A review of the wide range of technologies that are in common use for fish screening is provided, including physical and behavioural screening technologies.

Effective screening must be targeted to the species and lifestages of fish that are to be protected. Given the diversity of screening applications and environments and the need to consider the protection of a much-enlarged list of fish species than perhaps in the past, the developer or operator is faced with a potentially bewildering array of options. This review will help to guide users towards current best practice to assist in the task of screen selection and specification. There remain a number of gaps, where the effectiveness of new techniques has not been fully evaluated or where methods for particular species or applications have yet to be identified or developed. Recommendations for further screen development and evaluation are made.

EXECUTIVE SUMMARY

Background and Aims

Intakes used for water supply and hydroelectric power generation may harm fish if not properly screened to prevent fish ingress and there is also a risk of fish injury at intake structures or gratings. The issues and their remedies were reviewed for the National Rivers Authority by Solomon (1992). Now, more than a decade later, there have been significant changes in the law relating to fish screening, in the regulatory framework, and in fish screening and diversion technologies. It is, therefore, timely to readdress the issues. The aim of this Guide is to provide a clear description of the legal responsibilities of operators of water intakes and outfalls and, from a review of current, worldwide examples, to present a synopsis of available methods, indicating those that are known to work best for different species and lifestages of fish in different situations.

Potential impacts of fish entrainment and impingement

The design, installation and operation of fish screens and barriers can add significantly to the capital and operating costs of facilities. It is important for operators to recognise the potential impacts on fish and fish communities, which justify the costs of the required mitigation measures, and consider undertaking cost-benefit analyses.

Migratory diadromous species are historically recognised as being at risk as they often have to pass numerous water abstractions, as well as weirs and other hazards, on their journeys between rivers and the sea. In recent years there has been increasing recognition of the risk of entrainment into water intakes of juvenile freshwater fish during their downstream dispersal phases. Unscreened intakes on water transfer schemes may cause the unwanted introduction of new species or different genetic stock. In estuarine and coastal waters, impacts can arise from refineries, docks, and shipping but especially from thermal power stations, which abstract large volumes of cooling water. Desalination plants may also be developed in the future.

Review of screening and guidance technologies

This section presents a review of the wide range of technologies that are in common use for fish screening, including both physical and behavioural types. For salmonids and larger fish there are six main types of physical screening techniques: 1) traditional passive mesh screens – the most common fish exclusion method; 2) vertical or inclined bar racks; 3) rotary disk screens – originally designed for sewage treatment works but with some merits for intake screening; 4) Coanda screens –wedge-wire spillway screens mainly for upland hydropower applications; 5) the ‘Smolt-Safe™’ screen – another type of spillway screen and 6) band- or drum-screens modified for fish return. For juvenile and smaller fish there are four main physical screen choices: 1) passive wedge wire cylinder (PWWC) screens – the most widely used method for juvenile and larval fish protection; 2) small-aperture wedge-wire panel screens; 3) sub-gravel intakes and wells – which use the riverbed as a filter; and 4) the Marine Life Exclusion System (MLES™) – a water permeable geotextile barrier currently being evaluated in the USA. Other physical screening technologies not currently available in the UK include the modular inclined screen which is a wedge-wire screen which is tilted up from the horizontal, the labyrinth screen, which is a compact arrangement of vertical bar racks arranged in chevron formation and the self-cleaning belt screen.

Behavioural technologies can be used where positive exclusion fish screening is impractical or as a supplement to more conventional screen types. The best of these can be >90% effective against certain species when designed correctly and operated in suitable environmental conditions. As they do not provide a guaranteed barrier to fish passage, they are often used in less critical applications or where the alternative is to have no screening. There are five main types that have been used within the UK, comprising: 1) louvre screens – a semi-physical barrier; 2) bubble curtains – the most basic behavioural barrier; 3) electrical barriers – e.g. the 'Graduated Field Fish Barrier (GFFB™)'; 4) acoustic fish deterrents – which exploit the hearing sensitivity of fish; and 5) artificial lighting – either to illuminate physical structures or as an attractive or repellent stimulus (e.g. strobe lights). Behavioural technologies that are not known to have been used in the UK include: 1) turbulent attraction flow – which mimics natural river turbulence to guide fish into bywash structures; and 2) surface collectors – a bypass system which is based upon the natural tendency of salmon smolts to migrate to surface layers. Outfall screening may also be required to protect upstream migrating species. There are two main techniques suitable for screening outfalls: 1) mechanical mesh or bar screens and 2) electrical barriers.

Performance Criteria

While behavioural screens are expected not to achieve a complete barrier to fish, there is a common misconception that all positive exclusion fish screens, provided that they are designed with the optimum mesh size and velocity conditions, are 100% effective. In practice, this success rate is seldom achieved. Inspection surveys frequently reveal faults in the operation or maintenance of even the best designed screening systems. These can however, all be overcome with appropriate monitoring and maintenance. The effectiveness of screen measures should reflect the level of risk to the fish stock or fish community. The Environment Agency has adopted a 'Risk Assessment' approach, by which the required performance criteria for a screen can be determined according to such factors as the sensitivity of the fish stock or community and the other cumulative impacts upon it (e.g. other abstractions from the same watercourse, barriers to migrations, etc.).

Designing for Performance

Effective screening must, first of all, be targeted to the species and lifestages of fish that are to be protected. This will determine the method best suited, the critical times of the year and the specific design details for the fish screen (mesh size, etc.). Seasonal events may allow more focus in the design. Swimming performance of a species is strongly influenced by the length of the fish and by water temperature. The required criterion is that the fish approaching an intake should be able to swim fast enough and for long enough to ensure their escape via the bywash or any other route provided to return them to the main river flow. No statutory limits on escape velocities exist at present within the UK and the onus on the operator is to provide a system that avoids injury to fish. The chief purpose in hydraulics design is to avoid high velocity 'hot spots' that might cause fish to be impinged on the screen resulting in death or injury.

Selecting the Best Screen

Given the variety of screening applications and environments and the need to consider the protection of a much-enlarged list of fish species than perhaps in the past, the developer or operator is faced with a potentially bewildering array of options. Table 6.1 provides a summary of techniques that, from current knowledge, are likely to provide suitable screening solutions for different applications/environments and for the various categories of fish of concern.

It is stressed that screening is not always the best solution. It may be more economic and/or protective to modulate abstraction to avoid seasons, days or times of the day when fish are most at risk.

Monitoring for screen effectiveness: Recommendations for further work

From the review of screening technologies presented in section 3 of this Guide, it is clear that many different approaches exist and that there has been much innovation in recent years. The development of new techniques reflects the need to provide cost-effective solutions to suit an ever-widening range of circumstances. In practice, comprehensive scientific testing can be very costly and it makes sense to first answer basic questions on effectiveness from soundly designed generic studies. The number of techniques now in use could create an almost unlimited agenda for testing in order to cover the different environments, species and lifestages and the possible combinations of techniques. The Guide considers where resources would be best directed towards generic research to meet current needs. Fish screens usually form only part of an overall fish diversion or protection system and it is the performance of the entire system that needs to be proven. A variety of test methods is described, which can be used to assess screening efficiency, as well as for monitoring detailed fish behaviour in front of screens and bywashes. These include fish capture methods, biotelemetry, video monitoring, tagging, hydroacoustics and float-tagging. In addition to any generic research needs, site-specific commissioning trials may also be required to show that screening measures perform satisfactorily.

Knowledge gaps and future research needs

Solomon (1992) made a number of recommendations in respect of fish screening. The present Guide provides an indication of progress made since 1992 and comments on the current relevance of any outstanding issues. Key points arising from Solomon's recommendations are:

- A database of abstractions now exists but there remains a need for details of fish protection stipulations and measures.

- No significant advance has been made to investigate distribution and dispersion dynamics of coarse fish to aid in sympathetic siting and operation of intakes and further research is required.

- There is a pressing research need to assemble life-history data sets for particular species to investigate population control mechanisms to assess impact of losses at various life stages. Benefits of such research would no doubt spill over into other areas of fisheries biology and management.

- Screen slot and mesh sizes suitable for different species and lifestages are currently being researched in the USA and any new data from those studies should be investigated before commissioning new UK work. As the PWWC screen is one of the most important screening techniques currently available, good information on these aspects is essential and work should be commissioned if data are not found elsewhere.

- As the main large-volume water abstractors, there remains a need to investigate potential impacts from power plant abstractions on fish entrapment at intakes, either through commissioned R & D or owner-funded studies. Future work should concentrate in particular on designated fish species, especially lampreys, on entrainment of fish eggs, larvae and fry that are usually not fully represented in power station sampling, and on other species of conservation interest such as sea trout, smelt and eel.

- Wherever possible, through legislative provisions or voluntary cooperation, owners should be encouraged to ensure protection of all life stages of fish. This may be best achieved through screening measures, or through temporal modulation of flow to avoid abstraction during periods of high entrainment risk.
- Further, generic scientific testing of behavioural fish barriers is recommended.

CONTENTS

SUMMARY **4**

EXECUTIVE SUMMARY **5**

1 INTRODUCTION **13**

1.1 Background **13**

1.2 Key drivers for broadening the requirement for fish screening **14**

1.2.1 The developing legislative framework 14

1.2.2 Broadening scope of species to be protected 17

1.2.3 Changing water resources perspectives 17

1.2.4 Establishing 'green' credentials 18

2 POTENTIAL IMPACTS OF FISH ENTRAINMENT, IMPINGEMENT & ATTRACTION TO OUTFALLS **19**

2.1 Entrainment & Impingement **19**

2.1.1 Diadromous Fish 19

2.1.2 Freshwater Fish 21

2.1.3 Estuarine & Marine Fish 23

2.1.4 Non-Indigenous Species 25

2.2 Effects of Outfalls **26**

2.2.1 Attraction of Fish to Outfalls 26

2.2.2 Losses from Fish Farms and Reservoirs 27

3 REVIEW OF SCREENING & DOWNSTREAM GUIDANCE TECHNOLOGIES **28**

3.1 Positive Exclusion Screening Methods for Salmonids and Larger Fish **28**

3.1.1 Traditional Passive Mesh Screens 28

3.1.2 Vertical or Inclined Bar Racks 30

3.1.3 Rotary Disc Screens 33

3.1.4 Spillway Screens 35

3.1.5 Band or Drum Screens Modified for Fish Return 40

3.1.6 Econoscreen 45

3.2 Physical Screening for Juveniles and Small Fish **45**

3.2.1 Passive Wedge-Wire Cylinder Screens 45

3.2.2 Wedge-Wire Panel Screens 50

3.2.3 Sub-Gravel Intakes and Wells 51

3.2.4 Microfiltration Barriers 51

3.3 Other Positive exclusion Fish Screens **56**

3.3.1 Barrier Nets 56

3.3.2 Modular Inclined Screen 57
3.3.3 Self-Cleaning Belt Screens 58
3.3.4 Labyrinth Screens 59
3.4 Behavioural Barrier and Guidance Methods 61
3.4.1 Behavioural Deterrents Background 61
3.4.2 Louvre Screens 61
3.4.3 Bubble Curtains 66
3.4.4 Electric Barriers 69
3.4.5 Acoustic Guidance 73
3.4.6 Light-based Systems 85
3.4.7 Velocity Caps and Other Flow Control Measures for Offshore Intakes 88
3.5 Other Behavioural Guidance Techniques 91
3.5.1 Turbulent Attraction Flow 91
3.5.2 Surface Collectors 92
3.5.3 Eel Bypasses 93
3.6 Outfall Screening 95
3.6.1 Introduction 95
3.6.2 Positive exclusion Screens 95
3.6.3 *Electric Barriers* 96
4 PERFORMANCE CRITERIA 98
4.1 How Effective Should a Fish Screen Be? 98
4.2 Risk Assessment for Fish Screening 98
4.2.1 In General 98
4.2.2 For Hydropower Sites 99
4.2.3 For Water Transfer Schemes 100
4.2.4 Under SFFA s.14 100
4.2.5 Under the Habitats Regulations 102
5 DESIGNING FOR PERFORMANCE 104
5.1 Timing of Fish Movements 104
5.2 Intake Velocities and Fish Swimming Performance 105
5.2.1 Salmonid Smolts 106
5.2.2 Salmonid Kelt 106
5.2.3 Other Freshwater Fish Species 106
5.2.4 Marine & Estuarine Fish 109
5.2.5 Channel Velocities and Approach/Escape Velocities 110

5.2.6 Advisory Escape Velocities for Fish Screens 110
5.2.7 Uniformity of Flow Conditions 111
5.3 Fish Behaviour in Front of Screens 112
5.4 Effect of Screen Angle to Flow 112
5.5 Selection of Mesh Aperture 113
5.6 Screening for Epibenthic Species 115
5.7 Bywash Design Criteria 116
5.7.1 Bywash Location 116
5.7.2 Entrance Design 116
5.7.3 Light and Visual Attributes 117
5.7.4 Bywash Conduits and Outfalls 118
6 SELECTING THE BEST SCREENING SOLUTION 119
6.1 The Selection Process 119
6.2 Multiple Solutions and Non-Screening Solutions 122
6.3 Costs of Different Screening Solutions 122
7 MONITORING OF SCREEN EFFECTIVENESS: RECOMMENDATIONS FOR FURTHER WORK 125
7.1 Introduction 125
7.2 Priorities for Generic Trials 125
7.3 Scope of Work and Costs for Generic Trials 127
7.3.1 Measures of Performance 127
7.3.4 Sampling Fish Post-Screening and in Bywashes 130
7.3.5 Observing Fish Behaviour 133
7.4 Site-Specific Commissioning Trials 134
8 KNOWLEDGE GAPS AND FUTURE RESEARCH NEEDS 136
8.1 Review of recommendations from Solomon (1992) 136
8.1.1 *“A national database on abstractions should be developed to include details of fish protection stipulations and measures actually fitted.”* 136
8.1.2 *“Staff should be provided with a concise legal summary of legislation pertinent to fish screening and enforcement and that there should be a broadening of existing legislation to include all types of abstraction and all species of fish.”* 136
8.1.3 *“Operators should be required to fit appropriate fish screens whenever possible on new and existing abstractions, subject to provisions of the law.”* 136
8.1.4 *“R & D should be commissioned to investigate the timing, mechanisms and extent of migrations of 0+ and older coarse fish to assist in better defining periods when abstraction might be stopped.”* 136

8.1.5 *“R & D should be commissioned to investigate distribution and dispersion dynamics of coarse fish to aid in sympathetic siting of intakes (including diurnal patterns, swimming depths, etc.).”* 137

8.1.6 *“R & D should be commissioned to investigate population control mechanisms in 0+ fish to assess impact of losses at various life-stages.”* 137

8.1.7 *“R & D should be commissioned to investigate screen slot and mesh sizes suitable for different species and lifestages.”* 137

8.1.8 *“R & D should be commissioned to investigate the extent of fish entrapment at intakes in England and Wales.”* 137

8.2 Additional Recommendations for R & D 138

8.2.1 Juvenile fish mortalities and injuries in hydroelectric turbines 138

8.2.2 Eel and lamprey screening and guidance methods 139

8.2.3 Behavioural Barriers in General 139

9 REFERENCES 141

10 GLOSSARY 151

1 INTRODUCTION

1.1 Background

In England and Wales there are some 48,000 water abstractions licensed through the Environment Agency, for potable and industrial water supply, irrigation, flood defence, hydroelectric power generation and other purposes. Almost a third of these draw from surface waters containing fish populations, which may be harmed if the intakes are not properly screened to prevent fish ingress. Water outfalls can also present a hazard, where upstream-migrating fish may be attracted to the discharge and accidentally enter a works, a hydroelectric turbine or a fish farm, with a resultant risk of injury or delay to their migration. The issues and their remedies were reviewed for the National Rivers Authority by Solomon (1992). Now, more than a decade later, there have been significant changes in the law relating to fish screening, in the regulatory framework and in fish screening and diversion technologies. The consequences of fish entrainment are also better understood, as are the potential risks to fish populations. It is therefore timely to readdress the issues. It is also our aim to provide a clear description of the legal responsibilities of operators of water intakes and outfalls and, from a review of current, worldwide examples, to present a synopsis of which methods are known to work best for what species and lifestages of fish in different situations. The range of applications considered is as comprehensive as possible and the review includes estuarial and coastal, as well as inland waters. It is appropriate from a conservation perspective to widen the scope of review even further to include other nektonic organisms – notably crustaceans – that are potentially vulnerable to entrainment. Where relevant, these too are considered.

In preparing this document, the authors have drawn on a wide range of resources, including library holdings, material sourced from the Internet, contacts with scientists and other specialists in the field and contacts with operators of different technologies The starting point was the Solomon (1992) report. Other key guidance works consulted were:

- *Fish Passes and Screens for Salmon*. Report of the Salmon Advisory Committee (1997),
- *Notes for guidance on the provision of fish passes and screens for the safe passage of salmon*. Scottish Office Agriculture and Fisheries Department. (Anon., 1995a).
- *A UK Guide to Intake Fish-Screening Regulations, Policy and Best Practice with Particular Reference to Hydropower Schemes* (Turnpenny *et al*, 1998).

The present document summarises and updates the relevant information from these documents.

Prevention is better than cure and at the start of this guide we would urge that full consideration be given by operators to possible alternatives to fish screening: for example, modifying or ceasing the operational regime of the facility at crucial times of the year/ day/ tide, etc.

It can prove more cost-effective, for example to modulate periods of abstraction so that the risk to fish is minimised, rather than invest in costly fish screening measures. For new projects, such matters must be adequately addressed by *early* consultation with regulatory authorities, fisheries specialists and other interest groups, so that each can understand the others' constraints. Satisfactory fish screening or acceptable alternative arrangements must be provided which adequately address the level of risk to fish populations. The Environment Agency seeks to encourage improved engineering design, e.g. of pumps and turbines, or of fish rescue systems. While innovation must be the key to dealing with ever more challenging issues associated with new types of facility, different species and juvenile lifestages, there is an urgent need to make robust scientific assessments of new techniques. In this Guide, we will attempt to identify and prioritise research needs and gaps in available techniques.

For a variety of legal and historical reasons, many of our existing licensed intakes and outfalls have no protection or inadequate protection relative to present day needs. When considering screening technologies, the ability to retrofit equipment to existing facilities is therefore an important consideration.

The present Guide was commissioned by the Environment Agency, English Nature and CCW as a step towards improving the regulation of intakes and outfalls within the wider context of multi-species conservation across all aquatic habitat types.

1.2 Key drivers for broadening the requirement for fish screening

1.2.1 The developing legislative framework

1.2.1.1 The Salmon & Freshwater Fisheries Act (SFFA) 1975 as amended by the Environment Act 1995

The hazards of water intake and outfall structures were recognised in the fisheries law of England and Wales more than 80 years ago, with the introduction of screening legislation in the Salmon and Freshwater Fisheries Act of 18th July 1923[1] (revised 1975) (Howarth, 1987). Section 14 of the Act deals with the obligation of the owner or occupier of an undertaking to fit and maintain approved gratings, while s. 15 gives powers to the regulating authority to fit and maintain gratings at its own expense (gratings here may be interpreted as any device that prevents the passage of fish into the intake: Howarth, 1987). The most recent changes to SFFA s. 14 & 15 emanate from the Environment Act 1995, which made the Environment Agency the statutory authority in England & Wales. Key changes made at this time were the inclusion for the first time of fish farm intakes and outfalls as regulated structures, and the relinquishment of any regulatory approval mechanism: the latter important change effectively placed the onus of proof of effectiveness onto the owner or occupier; also, it became a requirement to provide a continuous bywash in any situation where screens are sited within a conduit or channel. The measures within SFFA s. 14 & 15 apply solely to the migratory salmonids, Atlantic salmon (*Salmo salar*) and sea trout (*Salmo trutta*) and technically apply to waters that are *frequented* by these species, a term which is interpreted to require demonstration that there is a self-supporting population of at least one of these species present, rather than one maintained by stocking. The Agency takes the view that this includes waters where there is a policy of reinstatement of migratory stocks.

The existing explicit law on fish screening may appear rather *Salmo*-centric in the present climate of fisheries and conservation, and even with regard to the protection of salmonids it is found to be wanting. Its powers do not for example extend to mills (including those operated as hydroelectric generation schemes) that have operated

[1] Schemes which have operated continuously since prior to this date have "licences of entitlement" and are exempt from provisions of the ACT.

continuously since prior to the 1923 Act[2]. Nevertheless, the Environment Agency and other regulators, including English Nature (EN) (for England) and the Countryside Council for Wales (CCW), have long held powers implicit within several Acts of Parliament that allow for the appropriate protection of any species of fish at water intakes and outfalls.

In England and Wales, powers emerge from the following statutes and amendments thereof:

1.2.1.2 The Wildlife and Countryside Act (WCA) 1981, as amended by the Countryside and Rights of Way Act (CRoW) 2000

The Wildlife and Countryside Act 1981 (WCA 1981) serves to implement the Convention on the Conservation of European Wildlife and Natural Habitats (Bern Convention) and Council Directive 79/409/EEC on the Conservation of Wild Birds (Birds Directive) in Great Britain. It is complemented by the Wildlife and Countryside (Service of Notices) Act 1985, which relates to notices served under the 1981 Act, and the Conservation (Natural Habitats, & c.) Regulations 1994. Amendments to the Act have occurred, the most recent being the Countryside and Rights of Way (CRoW) Act 2000 (in England and Wales only). There is also a statutory five-yearly review of Schedules 5 (protected wild animals, including fish), undertaken by the country agencies and co-ordinated by the Joint Nature Conservation Committee. The Act, amongst other things, makes it an offence (subject to exceptions) to intentionally kill, injure, or take any wild animal listed in Schedule 5, although the accidental killing or injury of fish through failure to provide adequate fish screening may not come into this category. The Act also provides for the notification of Sites of Special Scientific Interest (SSSI) – areas of special scientific interest for their fauna or flora.

Amendments to the Act under CRoW place a duty on Government Departments and the National Assembly for Wales to have regard for the conservation of biodiversity and maintain lists of species and habitats for which conservation steps should be taken or promoted, in accordance with the Convention on Biological Diversity. Schedule 12 of the Act strengthens the legal protection for threatened species and requires the regulator to submit a formal notice to the relevant nature conservation agency if the activity to be granted a permission constitutes an "operation likely to damage" (OLD) the SSSI (whether within or outside a SSSI). This in particular could influence the granting of consents in relation to intakes and outfalls.

1.2.1.3 The Water Resources Act (WRA) 1991

Many abstractions that have been licensed in recent years have, irrespective of the limited powers of SFFA s.14, been required to fit fish screens as conditions attached to new abstraction or impoundment licences under the Water Resources Act, s. 158. The Environment Agency in this way exercises its statutory duty under s. 114 of the Act to 'maintain, improve and develop salmon fisheries, trout fisheries, freshwater fisheries and eel fisheries'. Such conditions may require not only the installation of screening systems at the owner's expense but also the installation of monitoring equipment and monitoring surveys where these may be required. In fact, the WRA provides for great regulatory flexibility in achieving the above statutory duty, for example through placing limits on the timing of abstractions to avoid critical fish migration periods (diurnal, tidal or seasonal) or

[2] But see Howarth (1987), p. 74, who proposes that this exemption applies only to the *occupier* of the mill, and not the owner.

through more complex formulae related to available flows and water levels. Operating conditions will normally be negotiated with the owners to achieve the most workable and effective solution.

1.2.1.4 The Conservation (Natural Habitats, & Conservation.) Regulations 1994

European Directives require the enactment of enabling legislation in the Member States prior to enforcement. The Habitats Regulations, as they are commonly known, provide for the application of the European Habitats Directive (92/43/EEC) in England and Wales.

The aims of the Directive are "…*to contribute towards ensuring biodiversity through the conservation of natural habitats and of wild fauna and flora in the European territory of the Member States to which the Treaty applies*" (Article 2.1); and

"…*to maintain or restore, at favourable conservation status, natural habitats and species of wild fauna and flora of Community interest*" (Article 2.2)

The Habitats Directive requires the protection of designated species and habitats within Special Areas of Conservation (SACs). In the case of fish, the Annex II species occurring in England and Wales whose conservation requires the designation of SAC sites include:

- River lamprey (*Lampetra fluviatilis*)
- Sea lamprey (*Petromyzon marinus*)
- Brook lamprey (*Lampetra planeri*)
- Atlantic salmon (*Salmo salar*)
- Bullhead (*Cottus gobio*)
- Spined loach (*Cobitis taenia*)
- Allis shad (*Alosa alosa*)
- Twaite shad (*Alosa fallax*)

Any plan requiring consents for the development of a new project or significant modification of an existing project that might impact an SAC site will be subject to the Habitat Regulations. It should be noted that this does not necessarily mean that the project must lie within the designated area of the SAC; if the project might indirectly impact the SAC, e.g. a pollution source upstream of an SAC or an abstraction on the migration path of a designated species attempting to reach the SAC, then it will also be subject to the regulations. In England and Wales, the Environment Agency, as the 'competent authority' with regard to water intakes and outfalls, can only agree to a plan or project having ascertained by an 'appropriate assessment' that there will be no adverse effect on the 'integrity' of the SAC. The appropriate assessment is thus precautionary, and does not include consideration of economic factors. Such decisions will be based partly on advice from other statutory consultees, including English Nature or the Countryside Council for Wales. If, following an appropriate assessment, the Agency cannot conclude that there will be no adverse effect on the integrity of the European Site (from the plan or project alone, or in combination with other plans or projects and in the context of prevailing environmental influences), alternatives and modifications to the plan would have to be evaluated. It may be appropriate to consider fish screening techniques for relevant sites at this stage, as a potential solution to achieve no adverse effect on integrity. The cost of installing such screening would be appraised with the overall aim of achieving no adverse effect, but in the context of alternative solutions and potential reasons of overriding public interest.

Interpretation of the Habitats Regulations is complex. Further information can be obtained from the Environment Agency via its website (www.environment-agency.gov.uk) and from the Joint Nature Conservation Committee (www.jncc.gov.uk).

1.2.1.5 The Environment Act (EA) 1995

This incorporates and amends the SFFA *s*.14 and 15 powers, as described above.

The Water Environment (Water Framework Directive) (England and Wales) Regulations 2003 &

1.2.1.6 The Water Environment (Water Framework Directive) (Northumbria River Basin District) Regulations 2003

Over-arching powers affecting the regulation of fish habitats will also emerge from the European Water Framework Directive (WFD) (2000/60/EC), introduced in the UK in December 2000, which is expected to be fully implemented in the UK by 2015. It requires all inland and coastal waters to reach 'good status' by this date. It will do this by establishing a river basin district structure within which demanding environmental objectives will be set, including ecological targets for surface waters. The above Regulations transpose the WFD into English and Welsh Law; the second set of Regulations is separated from those pertaining to the main body of England and Wales as the Northumbria River Basin District contains a small part of Scotland.

The full implications of the WFD with respect to fish screening have yet to emerge but it is clear that the requirement to achieve demanding ecological status objectives will strengthen rather than weaken existing powers. Essentially it is likely to mean that owners will be required to ensure that any new developments meet sustainability criteria and that they do not lessen the existing ecological status of a water body but if anything improve it.

1.2.2 Broadening scope of species to be protected

The consideration of species other than migratory salmonids goes beyond the statutory remit arising from the Habitats Directive. It comes from a modern perspective on the merits of holistic ecological conservation and the increasing recognition e.g. that many species of freshwater fish are much more mobile within the river continuum than was once understood. Thus the within-river ('potamodromous') movements of non-migratory trout and of coarse fish are now recognised, in addition to the more traditionally known migrations of diadromous species, such as salmon, eel, lamprey and shad, as are the migrations of euryhaline species such as smelt (*Osmerus eperlanus*), flounder (*Platichthys flesus*) and grey mullets. Any of these concerted movements or migrations within species put them at increased risk of entrainment. In some species, the risk of losses due to entrainment exacerbates an already declining trend in the status of stocks, for example in the eel, or may slow down the rate of recovery of returning species, such as smelt or shad.

1.2.3 Changing water resources perspectives

Recent years have seen an upsurge in interest in renewable energy development, including hydroelectric power (HEP) projects. HEP schemes generally use relatively large proportions of a river's flow (sometimes as high as 95% of the dry weather flow), which makes provision of effective fish screening a particular challenge. On some rivers the risk to fish populations is compounded by the cumulative effect of the need to pass two or more HEP schemes in succession. Risks arise from injury to fish in the turbines where they are able to pass through, from impingement on the screens where they are inadequately designed or operated or from delays to migration – with possible increased predation risk - where bywashes are not readily found by the fish (Turnpenny *et al.*, 1998, 2000). Anyone interested in this subject is also advised to refer to the Agency's publication "Hydropower – A handbook for Agency Staff" (May, 2003), available from Environment Agency offices.

1.2.4 Establishing 'green' credentials

A positive factor in fish screening is that many operators of intakes and outfalls want to be seen to be environmentally responsible and will do more than the law requires, provided that this does not impact too heavily on the viability of their operations. For example, a number of water companies who operate under 'licences of entitlement' or abstract from rivers not frequented by migratory salmonids have voluntarily provided fish screening. This is often to establish good relations with anglers or to meet internal environmental management objectives.

2 POTENTIAL IMPACTS OF FISH ENTRAINMENT, IMPINGEMENT & ATTRACTION TO OUTFALLS

The design, installation and operation of fish screens and barriers can add significantly to the capital and operating costs of facilities. It is important for operators to recognise the potential impacts on fish and fish communities, which justify the costs of the required mitigation measures.

2.1 Entrainment & Impingement

2.1.1 Diadromous Fish

Diadromous fish are migratory species that move between the sea and freshwater, or vice versa. These may be subdivided as follows (McDowell, 1988):

- *Anadromous*: spending most of the life in the sea but moving into freshwater to spawn (e.g. salmonids, shads, smelt, lampreys);
- *Catadromous*: spending most of the life in freshwater but moving into the sea to spawn (e.g. eels);
- *Amphidromous*: marine or freshwater species which spend a significant proportion of their life in the other (freshwater or marine) phase but not for the purpose of spawning (e.g. bass, but also numerous marine species whose early juvenile life is spent above the salt wedge in estuaries, e.g. sole, *Solea vulgaris*, see Coggan and Dando, 1988).

These migratory species are notably at risk as they will often have to pass numerous water abstractions, as well as weirs and other hazards, on their journeys between rivers and the sea. It is owing to the compounding risk of these multiple hazards that migratory species were the first to be explicitly protected in law. Although this explicit protection did not extend to the non-salmonid species prior to the introduction of the Habitats Directive, the Environment Agency (and NRA previously) has used its powers under the Water Resources Act to place protective conditions on abstraction and impoundment licences and land drainage consents.

While it might be tempting to assume that the salmonids are adequately catered for within present fisheries law, there remain many unscreened intakes and outfalls on salmonid rivers throughout England and Wales. A number of reasons account for this:

- Most commonly, the intakes/outfalls were built prior to the introduction of the 1923 Act and were not required to be screened.
- In other cases, salmonids were not present in significant numbers at the time when the intake/outfall was constructed (usually as a result of industrialisation) but have recovered or been reintroduced in later years (e.g. rivers Thames and Trent and many rivers in Wales, the North-East and North-West).
- In the case of fish farms, legislation was introduced only with effect from 1st January 1999.
- Subject to the Environment Agency’s SFFA s.14 Risk Assessment procedure, intakes/outfalls where the risk is evaluated to be negligible may, at the Agency’s discretion, be exempted from fish screening.

- The operator may not be complying with the requirements of SFFA s.14 under Agency policy.
- Predecessor agencies may not have enforced their powers rigorously.

There is little quantitative information on the risk to salmonids from unscreened intakes. The cooling water intakes of estuarine power stations have often been deemed a likely threat to salmon and sea trout runs but this has not generally been borne out by survey data from English and Welsh stations. Solomon (1992) refers to catch rates of up to 10,000 smolts per annum at Uskmouth A & B stations (R. Usk estuary, S. Wales). Records kept by Uskmouth B Power Station from the 1960s until its initial closure in 1994, recorded an annual impingement rate of 189 smolts (range 22-493: Fawley Aquatic Research, unpublished data), suggesting that it was mainly the A station or the combination of the two that was responsible; by comparison, the wild smolt population in the R. Usk was estimated to have varied between 8.4×10^4 and 3.0×10^5 over the years 1962-1987 (Aprahamian and Jones, 1989), indicating a loss rate of around 0.1% for the B station alone. Fawley Power Station (Hampshire), which abstracts from the migration path into the R. Itchen and R. Test, caught an estimated 203 salmon smolts and 42 sea trout smolts during a survey conducted from March 1978-March 1979, and records for the newly constructed Shoreham Power Station (R. Adur estuary) during three years of post-commissioning trials (2001-3) show an average catch of 18 sea trout smolts per year (Fawley Aquatic Research, unpublished data). An earlier survey at Fawley conducted daily from February 1973 to January 1974 found no impinged salmon smolts and 41 sea trout smolts (Holmes, 1975). On river intakes, where migratory fish may be forced to pass closer to the entrances, significantly larger proportions could become entrained in the absence of any screening. Solomon (1992) refers to a count of 1059 smolts entrained at a Hampshire fish farm intake, which was estimated to be around 5% of the run at that point. A model of smolt entrainment relative to potable water abstraction on the R. Thames indicated that, in wet years, about <5% of the sixty-thousand salmon smolts stocked annually might be entrained into water supply intakes, and around 15% in years of moderate rainfall. During extreme drought, it was predicted that this figure could rise to ≥80% (Solomon, 1992). As a result of this, Thames Water Utilities voluntarily installed acoustic fish deterrents on all of their intakes during the mid-1990s.

For the other migratory species, losses for the most part are not well quantified. While eels and elvers have been recorded as entrained at many water intakes around the country, there has been no concerted effort to quantify the impact of entrainment on stocks at a regional or national level. With recent evidence of the sharp decline in eel stocks internationally, and the conservation focus on lampreys, this has now become a concern. Perhaps the most difficult issue with these species is the potential conflict with run-of-river hydroelectric schemes. Being fine-bodied in cross-section, individuals of much greater length than a salmon smolt can pass through a conventionally sized smolt screen and will have a higher risk than a smolt of being chopped by a rotating turbine runner. For example, a conventional ½ inch (12.5 mm) square-mesh smolt screen will stop smolts of ≥12 cm in length and eels of ≥36 cm in length (Turnpenny, 1981): passing through a turbine, the eel would be three times more likely to be struck by the turbine runner. These species also seem less amenable to behavioural guidance methods. This therefore represents one of the major fish screening challenges as we look increasingly towards renewable energy sources.

An indication of the potential for lamprey entrainment is given by the results of an entrainment survey conducted by the former National Rivers Authority (NRA) at

Yorkshire Water's Moor Monkton pumping station on the R. Ouse (Frear and Axford, 1991). Over a 15 month period between January 1990 and May 1991, over 16 thousand lampreys were impinged. Most were recently metamorphosed down-migrating river lampreys (known as 'transformers'), along with some brook lamprey transformers, averaging about 100mm in length. The entrainment rate was very sensitive to the proportion of river flow abstracted. This risk has now largely been eliminated through new fine-screening measures (see section 3.2.1).

The shad species, of which the twaite shad greatly outnumbers the allis shad, also represent a significant challenge. Unlike the salmonids, juveniles return to sea at the end of their first summer. At this stage they are typically around 6 cm in length, compared with 12-25 cm for a 1-2 year old smolt, and will not be excluded by a smolt mesh. They also possess loosely attached scales and are particularly sensitive to any form of mechanical contact. As they migrate seawards in shoals they can be entrained in considerable numbers during the autumn migration period. Records from the Hinkley Point A & B Power Stations (Somerset) between 1981 and 2001 show catches on the cooling water intake screens of up to 42 juvenile shad in a 6-hour sampling period (Fawley Aquatic Research Laboratories Ltd, unpublished). The highest estimated annual catch (in 1990) was 17,000 shad, mostly 0-group.

2.1.2 Freshwater Fish

Little information on the entrainment and impingement of non-migratory freshwater fish was available in this country before the 1990s, primarily owing to a lack of interest. It was feared from overseas studies that large quantities of coarse fish fry were likely to be entrained at freshwater intakes. Hadderingh (1982) had reported daily entrainment rates of up to 25 million coarse fish fry during the peak season (May) at a coal-fired power plant on Lake Bergum in Holland.

Solomon (1992) described work carried out by Thames Water at Walton-on-Thames water treatment works (WTW) in the late 1980s. The project was intended to look at smolt entrainment and coarse fish fry were not fully quantitatively sampled. Solomon estimated that for the 1989 season, the numbers of 0+ fry entrained (excluding pinhead fry) might lie somewhere between 8.7×10^5 and 2.9×10^6, with ~20,000 fish of age 1+ being entrained in the same year. Fry appeared in samples from the end of May onwards, at lengths of 23-30mm, the largest growing to about 70 mm by the end of the season. The pattern observed elsewhere in other European rivers (e.g. Pavlov, 1989) is that newly hatched pinhead fry first occupy sheltered areas on gravel beds or in vegetation at the margins of rivers. Soon, they inflate their swimbladders and become buoyant, allowing them to be carried and redistributed downstream by currents. It appears to be at this time that they become most vulnerable to entrainment. Peak catches are generally reported at night and during periods of high flow.

Other freshwater fish entrainment and impingement work has since been carried out at UK power stations, including Didcot (R. Thames) and Ratcliffe-on Soar (R. Trent) and at Farmoor WTW (R. Thames). The Ratcliffe work was reported in detail by Smith (1998). Smith recorded peak fry capture rates in June and July of 10 fry per hour in daylight, rising to a maximum night-time rate of 4,500 per hour. Annual fry entrainment rates at the plant of $3.45\text{-}7.98 \times 10^5$ were recorded between the years of 1994 and 1997 (Table 2.1). The dominant species were roach (*Rutilus rutilus*), bream (*Abramis brama*), bleak (*Alburnus alburnus*) and chub (*Leuciscus cephalus*). Smith used an Equivalent Adult (EA) procedure (Turnpenny, 1988; Turnpenny and Taylor, 2000; Turnpenny and

Coughlan, 2003) to estimate the implied loss to the adult stock. The EA is a useful means of representing losses of fish of mixed ages by computing survival trajectories through to a standard age (the nominal age at first sexual maturation) based on life history data; it should be noted that it does not take account of possible density dependent factors that may operate within the population dynamics of a species. Table 2.1 shows the entrainment losses also expressed in EA terms.

Table 2.1 Estimated entrainment rates of coarse fish fry at Ratcliffe-on-Soar Power Station (R. Trent) and their Equivalent Adult numbers (after Smith, 1998).

Year of Sampling	1994-5	1995-6	1996-7	Average
Numbers of fry entrained	798,000	345,000	729,000	624,000
Numbers of Equivalent Adults	3,940	1,460	1,480	2,290

Smith (1998) estimated the loss to represent 4.1% of the stock size within the impounded reach.

Entrainment on the upper R. Thames was examined by Turnpenny (1999) on behalf of the Environment Agency. A potable water intake located at Farmoor (near Oxford) was monitored continuously between the months of June to August 1998 following the installation of an acoustic fish deterrent (AFD) system. It was found to entrain few coarse fish fry compared with the Walton intake: only 246 fry were entrained over the whole period, of which 80% were perch (*Perca fluviatilis*). This marked contrast with the Walton situation was attributed to a number of factors:

- a cold spring in 1998 led to poor spawning success and surveys of the river in the locality of the WTW showed overall low densities of fry to be present;
- intake velocities at Farmoor are very low compared with Walton (approximately 0.08 ms^{-1}, *cf.* 0.71 ms^{-1} at Walton);
- the AFD system was operated on alternate days for the testing and fry entrainment was reduced by 87% on days when the AFD was active.

Another Thames study reported in Turnpenny (1999) was carried out as part of the National Power/ PowerGen Joint Environmental Programme in May-June 1992, when samples of water were tapped off from the cooling water system of Didcot Power Station. The water was passed through a plankton net, allowing very small fry to be retained. Ten species of coarse fish fry were captured, starting at a length range of 8-13 mm in late May and reaching 12-26 mm by July. This work suggests that the Walton study may have missed a significant proportion of smaller fry, which is not surprising given that the sampling was aimed primarily at salmonids. The true catch rates at Walton may therefore have been considerably higher than the findings given above would suggest. The entrainment rate at Didcot was estimated at 1.2 $x10^3$ fry per day over this period. This yielded an annual entrainment estimate of around 1.9 x 10^6 fry, in this case including pinheads. However, Smith (1998) considered this likely to be an overestimate, perhaps by as much as a factor of ten, as it assumed uniform entrainment over the season, whereas entrainment tends to be patchy with time and decline over the season as the fish grow and are more able to resist entrainment.

The overall purpose of Turnpenny's (1999) Thames study was to assess the potential impact of entrainment from water abstractions on adult fish populations in the lower freshwater Thames (Hurley to Teddington). Based mainly on the Walton data, an estimate was made of the potential loss of fish if all the abstractions within this reach were operated at their maximum licensed capacity. It was shown that, in Equivalent Adult terms, the numbers of fry entrained per annum could equate to around 45% of the adult stock. While continuous operation of all of the intakes at their licensed limit is never likely to happen, and the figures are known to be very imprecise, the findings confirm that it is an issue that cannot be ignored, there or in other heavily abstracted rivers.

Land drainage pumping schemes are susceptible to fish entrainment issues, particularly where accumulations of fish overwinter in unscreened intake chambers. Fisheries staff in the Anglian Region of the Environment Agency have on occasions been called out to fish rescue operations in which tens of thousands of juvenile coarse fish have been recovered.

From the information reviewed on freshwater fish entrainment, benthic species such as bullheads and loaches seldom occur in survey lists. Presumably their benthic habit ensures that they are out of the reach of most types of intake and so are protected from the risk of entrainment.

2.1.3 Estuarine & Marine Fish

The main abstractors from estuarine and coastal water are thermal power stations, so located principally to take advantage of the large volumes of cooling water available. Intake flow rates for directly cooled generating plants are usually between 0.5 to 5 x 10^3 megalitres per day (Turnpenny and Coughlan, 2003). Other estuarine and coastal cooling water users include refineries and shipping. Various port and harbour operations require pumping of water, for example emptying of dry docks and operation of shipping locks, all activities which may incur fish mortalities. In other countries, tidal power generation and desalination plants may be added to this list but, while planned for the UK, none are known to the authors to date.

Much research into the effects of power station water use was carried out by the Central Electricity Generating Board (CEGB) of England and Wales and its successors, mainly prior to the privatisation of the industry in the late 1980s. An up-to-date summary of their research into the ecological effects of cooling water abstraction is given by Turnpenny and Coughlan (2003). Table 2.2 summarises the quantities of fish impinged on the cooling water intake screens of various stations that were monitored from the 1970s onwards, revealing that quantities amounted to tens of tonnes per annum in some cases; in the case of the French station at Gravelines, hundreds of tonnes. It is notable that the highest flow-standardised catch rates mostly occur at stations sited on the open coast. The data shown are raw catches, not equivalent adult-adjusted figures, and do not include entrained[3] fish eggs and larvae. For most stations, entrainment data were not collected but some idea may be gained from very detailed studies that were undertaken at the Sizewell A and B power stations over more than a decade (Turnpenny and Taylor, 2000): here, equivalent adult values calculated for impinged-plus-entrained fish of commercial species were higher by a factor of 14 in 1981-2 surveys and by a factor of 21 in 1992. The commercial value of fish lost to impingement and entrainment (as adult

[3] The term 'entrainment' refers to fish (including eggs and larvae) being drawn into the plant, as opposed to those that become impinged on filter screens.

equivalents) at the Sizewell A+ B sites was estimated at £0.5 million per annum (1994 Lowestoft market prices). While these figures may be seen as significant, it was also shown that the catches were minor relative to wastage in the fishing industry, for example several orders of magnitude lower than bycatch mortalities in the Wash shrimp fishery.

Table 2.2 Estimated annual fish impingement catches at various UK and French power stations (after Turnpenny and Coughlan, 2003).

Power Station	Location	Total quantity of impinged fish (tonnes/yr)	Catch per unit of CW flow (kg/10^6 m^3)
Dungeness A	Open coast	93	190
Sizewell	Open coast	43	73
Gravelines	Open coast	240	48
Dungeness B	Open coast	20.6	40
Hinkley B	Estuary	24	31
Dunkirk	Estuary	13	19
Fawley	Estuary	6.4	19
Wylfa	Open coast	2.4	5
Kingsnorth	Estuary	6.6	4.4
Shoreham	Estuary	0.68	3.8

Although the power industry sought to put these catch figures into the perspective of other causes of fish mortality, the overriding aim of the industry has been to improve intake technologies so as to reduce catches. Considerable progress was made and continues to be made in this regard. This point is illustrated by the history of the Sizewell sites (Turnpenny and Taylor, 2000). Prior to the construction of the Sizewell B pressurised water reactor, local fishermen lodged objections to the new plant on the grounds that fish stocks might be adversely affected. The fish catches at the A-station were monitored and quantified (see below) and much work was done to improve the intake design to reduce the fish catch. Design changes included:

- reduction of intake velocities;
- fitting a velocity cap to the intake to eliminate vertical flow components (see section 3.4.7);
- elimination of any intake superstructure (which tend to act as artificial reefs that attract fish);
- location of the intakes further offshore where juvenile densities are lower;

- installation of a fish return system.

Fish return systems collect the fish backwashed from the screens and return them to the wild. They employ various ‘fish-friendly’ design features that reduce the handling damage to fish (see section 3.1.5).

Table 2.3 shows for the key commercial fish species the benefit of these improvements. It is seen that for flatfish and cod, catches are reduced by >90% relative to the A-station for a unit of flow. Herring benefited least, being too delicate to be handled safely by the fish return system.

Table 2.3 Catch of fish per unit of cooling water flow at Sizewell B Power Station relative to catches at the A-station which does not have the intake design improvements (after Turnpenny and Taylor, 2000).

Species	B-Station / A-Station Mortality Rate		
	Due to Intake Design & Location	Due to Fish Return System	All Measures
Plaice	54%	0%	0%
Sole	63%	4%	3%
Cod	100%	6%	6%
Dab	46%	20%	9%
Bass	91%	11%	10%
Whiting	79%	52%	41%
Herring	74%	100%	74%

At newer estuarine/coastal power stations, additional improvements have been introduced, principally in the form of acoustic fish deterrent (AFD) systems (see section 3.4.5). AFD systems are especially effective against delicate pelagic species such as herring, sprat, smelt and shads and therefore complement fish return technology. Shoreham Power Station (Sussex), commissioned in 1999, incorporates AFD and fish return techniques and, as is seen from Table 2.2, has exceptionally low fish catch. The same technology pairing has recently been introduced at the new Great Yarmouth Power Station (Norfolk) and at Fawley Power Station (Hampshire). This is further discussed in section 3.

2.1.4 Non-Indigenous Species

Pumping large volumes of water from one location to another potentially promotes the risk of spreading invasive, non-indigenous species around and between catchments. Zander (*Stizostedion lucioperca*) provide an example of this. Having been introduced into the Fens in East Anglia many years ago, they have progressively spread into the river and canal systems of the Midlands by various means. British Waterways, operators of the canal systems, are concerned to avoid exacerbating the spread of the species via any of their water transfer schemes. For example at Napton in Warwickshire, a pumping station was installed some years ago to recharge a flight of locks. This involves pumping water from the bottom of the flight to the top, which is located within an adjacent watershed. Following advice, the pump entrance was retrofitted with a two-tier bubble curtain system, designed to reduce the risk of zander eggs and fry present in the downstream catchment from entering the pumps. A similar approach used in the Eastern Block was reported by Pavlov (1989), where 80% deflection efficiency was achieved with a single-tier bubble curtain. Zander have subsequently been found in the adjacent catchment but, as with the spread of any alien species, this may have been from one of a number of causes, such as angling activity, aquaculture, transfer by birds, and so on. The level of impact arising from catchment transfers by water pumping is therefore impossible to quantify but prevention of cross-catchment contamination through use of positive exclusion screens with appropriately small apertures is to be recommended.

2.2 Effects of Outfalls

2.2.1 Attraction of Fish to Outfalls

Seasonal upstream migrations are undertaken by a wide variety of species, including the spawning runs of salmonids, shads, smelt, lampreys and many coarse fishes, as well as the migrations of elvers towards river and lake habitats. Rheotactic reactions render these lifestages vulnerable to attraction into outfalls from hydroelectric schemes, fish farms, industrial waste discharges or other sources. Not only is this a distraction, which can delay upstream progress, they may also risk injury by attempting to enter the discharges. Distraction can be a particular problem where a discharge occurs near to the bottom entrance of a fish pass, and care must be taken to ensure that the discharge reinforces, rather confuses, the attraction flow from the pass.

Solomon (1992) conducted a questionnaire survey among NRA Regional Fisheries Officers to investigate the size of this problem. Three out of ten of the former NRA regions reported a ‘significant’ problem and four others a ‘minor problem’. Solomon estimated that there were a ‘few tens’ of problem sites throughout England and Wales. Most of the problems related to adult salmon or sea trout entering discharges, e.g. through poorly maintained screens. Where this is notified to the Environment Agency, the fish can be rescued by electrofishing. At a fish farm on the Hampshire Avon, such a rescue operation for adult salmon revealed that many coarse fish had also entered the outfall.

Solomon (1992) explained the significance of the problem in terms of possible truncation of the spawning distribution, increased risk of illegal exploitation of trapped fish and restricted upstream angling opportunities. Adult fish frustrated in their attempt to ascend the river may also injure themselves when attempting to pass outfall screens where there is no alternative route available to them.

It is important that outfall screening remains effective even when the outfalls are not discharging. Otherwise, fish may enter them during quiet periods, subsequently risking becoming trapped or injured. Hydroelectric tailraces, which often operate intermittently, are prone to this problem. A run-of-river plant at Beeston on the River Trent, has electric outfall screens. When initially commissioned, the power to the screens was switched off when the plant shut down. After two large bream were found severed below the plant, probably as a result of collision with the runner blades, the operating regime was altered to keep the electric barrier continuously energised. Since that time no further problems have been reported (Fawley Aquatic Research, unpublished report).

There are similar, anecdotal reports of adult salmon mortalities at other British hydroelectric schemes where the velocity of flow exiting the turbine draft tube has been relied upon as an alternative to screening. It appears that actively migrating fish become attracted to the residual flow emanating from the draft tube when the turbine is not running. At start-up, they then become at risk of blade strike from the turbine.

2.2.2 Losses from Fish Farms and Reservoirs

Amendments under the Environment Act 1995 to *s*.14 of SFFA 1975 were partly in recognition of the potential environmental impacts caused by losses from fish farms and stocked reservoirs when inadequate outfall screening allows fish to escape. Losses may be of indigenous species of different genetic origin than the native river stock, but often involve non-indigenous fish such as rainbow trout (*Oncorhynchus mykiss*). Problems most often occur during flood events when reservoirs overtop unscreened spillways or when fish farm outlet screens are damaged or not set sufficiently high to cope with flood water levels. Fish stocks in the rivers Exe in Devon or Test in Hampshire, and others that support large numbers of fish farms, are heavily contaminated by rainbow trout, to the potential detriment of native species.

3 REVIEW OF SCREENING & DOWNSTREAM GUIDANCE TECHNOLOGIES

This section presents a review of the wide range of technologies that are in common use for fish screening. Where experience allows, best practice is identified. Later sections discuss where the different techniques may be of benefit. More promising 'cutting-edge' techniques are also described, some of which are still under development or may require further evaluation.

3.1 Positive Exclusion Screening Methods for Salmonids and Larger Fish

3.1.1 Traditional Passive Mesh Screens

Static screens constructed of mesh are presently by far the most common method of fish exclusion. A standard smolt-screening arrangement, as found at many hydroelectric stations, as well as drinking water and industrial water supply intakes, uses flat panels of mesh, fixed to a stiffening frame (Plates 3.1 & 3.2) (Aitken *et al*., 1966). One or more such panels are inserted into vertical slots in a fixed supporting structure, which may have an overhead walkway and lifting gear to facilitate removal and replacement of individual panels for cleaning and maintenance. Alternatively, the panels can be made to pivot, so that debris can be back-washed off by the water flow, but this may lead to a risk of fish passing through while the screens are being turned and is therefore not ideal. Suitable systems can be designed for any size and most configurations of intake. Ideally, the screen should be aligned flush with the riverbank, or else at an angle to the flow to assist in guiding fish towards a bywash positioned at the downstream end of the screen. The angle is calculated such that the flow vector normal to the screen face is below the required escape velocity for the target fish species and sizes (see Section 4). The size of individual panels used is determined by the overall screening area and by practical considerations of handling.

Plate 3.1 Fixed panel screen installed on the R. Afan, Port Talbot in 2003, to prevent salmonid smolts from entering the docks feeder.

The mesh can be made from one of a number of materials, including plastics but stainless steel is the norm. The ease of cleaning and extended life expectancy outweigh the initially higher capital costs. Weldmesh is easier to clean and cheaper to produce than e.g. a woven mesh. Plastic meshes, used on the band- or drum-screens of some continental power stations, are probably not sufficiently robust for use in open water. Stainless-steel wedge-wire is very effective, particularly where it is required to exclude juvenile fish also (see section 3.2.2). The amount of debris reaching the fish screen may be lessened by placing a coarser trash rack in front of it, without affecting fish passage. In this case a bywash entrance must also be provided upstream of the trash rack as well as by the smolt screen, so that larger fish (e.g. kelts) can bypass the structure. An example of this is found at Scottish Hydroelectric's Dunalastair Dam (see Aitken *et al.,* 1966). Alternatively, a larger bar spacing can be used (e.g. 15 cm) or gaps can be left at intervals to allow larger fish to pass.

Plate 3.2 Fixed panel screen installed on the R. Plym, Devon in 2003, protecting the entrance to an industrial supply offtake.

Selection of a suitable mesh aperture for a standard square mesh screen design is discussed in section 5.7.

3.1.1.1 Design Best Practice for Panel Screens

The main design requirements are as follows:

1 the mesh size should be selected to ensure exclusion of the minimum target fish size based on preventing penetration of the fish's head (see equation 1);

2 the screen should be flush with the riverbank for a lateral river intake or, when placed across a channel, angled (in plan view) relative to the channel to guide fish into a bywash;

3 a suitable bywash should be provided where the screen is placed across the channel;

4 the velocities ahead of the screen should be low enough to allow fish to escape without injury.

For shallow water applications (typically <1 m), it may be practicable to operate fixed screens with manual raking or brushing. For deeper water designs, screens will need to be removable for cleaning. For this purpose they are normally dropped into vertical slots from which they can be hoisted out for cleaning. In this case best practice will also include:

- provision of two sets of slots, one behind the other, allowing a cleaned screen to be inserted before the soiled screen is removed;
- provision of adequate seals around the screen to prevent fish passage or injury;
- provision of a datum mark on the screen which aligns with a mark on the slot rails to show when the screen is fully seated and sealed.

3.1.1.2 Applications

Passive mesh panel screens are suitable for a wide range of applications provided that the above criteria are met. Limiting factors may include required frequency of cleaning in order to avoid risk of blockage, structural strength in relation to flood damage risk and hydraulic head loss where small mesh sizes are used. These factors tend to become more significant with larger abstraction flows. Factors favouring this type of screen include no necessity for electrical power and relatively low capital cost, especially on small installations.

3.1.1.3 Fish Species/Life Stages

Suitable for all fish species and life stages, subject to meeting design requirements 1-4 above.

3.1.1.4 Ease of retrofitting

Retrofitting of this type of screen is possible but depends largely on site characteristics. A common problem with retrofitting fixed fish screens as a replacement for simple trash racks is that water velocities may be too high. In this case it may be necessary to widen or deepen the intake.

3.1.2 Vertical or Inclined Bar Racks

Vertical or inclined bar screens have in the past been used mainly as trash racks for debris exclusion; many are fitted with moveable tines or raking systems that keep the screens clear of rubbish. Back- and front-raked systems are available, but the former necessarily lack horizontal braces and are not recommended for closely-spaced bars. Conventional trash rack spacings may be anywhere between 38 mm and 150 mm, depending on the application. Inclining the bars by 10^{o} to the vertical assists in maintaining the weight of the rake against the bars. Mild steel (with or without galvanization) is commonly used as the construction material, or stainless steel. One north-American manufacturer[4] offers robust plastic trash racks.

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[4] (Structure Guard Inc., Maine, USA www.structureguard.com)

Currently there is interest in replacing two-tier systems comprising trash racks followed by mesh fish screens with a single-tier bar rack using a smaller bar spacing that can act as a fish screen. This provides a self-cleaning alternative to the traditional manually cleaned mesh panel screen described above. Scottish and Southern Energy plc (SSE) have been investigating this approach for a number of years. Clough *et al.* (2000 unpublished) undertook controlled trials in a test flume constructed within a large fish pass at Gaur (R. Tummel system), in which hatchery smolts were released upstream of test bar racks of 10 mm or 12.5 mm spacings. The aim was to assess whether the risk to fish was any different when compared with a regular 12.5 x 25 mm steel mesh screen. The screens were presented either at 75^{o} or parallel to the flow and provided with an adjacent bywash at approach velocities of up to 0.4 ms^{-1}; bywash entrance velocities were 1.5 times the approach velocity. Fish reactions were observed by video camera, using infra-red lighting at night. It was concluded that:

- there was no evidence that 10 or 12.5 mm spaced bar screens were any more likely to impinge smolts than the 12.5 x 25 mm mesh traditionally used: no fish were impinged on either type during the tests;
- smolt behaviour was similar for both screen types, irrespective of screen alignment and whether light or dark, at all velocities tested.

Plate 3.3 Experimental apparatus used in bar rack trials at Gaur (Clough *et*. 2000).

In 2001, SSE installed vertical bar screens at a small hydroelectric intake. Bars were spaced at 12 mm and cleaned automatically by a raking machine once a specified pressure differential was measured across the screen. This has performed satisfactorily, except under extreme flow conditions when the raking system has been hampered by gravel shoals accumulating at the base of the screens (Dr Alasdair Stephen, SSE, personal communication. SSE plans to install a raked bar screen for smolt exclusion on one of its larger schemes in 2004.

Other UK examples of automatically raked bar screens designed specifically for fish exclusion include a pair of screens with 10 mm spacings located at a small (2 m^3s^{-1}) hydroelectric intake at Dolanog on the R. Vyrnwy in Wales, and a larger 15 m^3s^{-1} capacity smolt screen at Innogy Hydro's recently refurbished Stanley Mills plant on the R. Tay (Perthshire). The Dolanog screens (New Mills Hydro Ltd) have individual hydraulically powered rakes; the river does not contain migratory fish at the abstraction and the screens are aimed primarily at excluding brown trout. The Stanley screen is a large angled bar rack, some 33 m long and 2.4 m deep, with a single travelling rake supported on an overhead rail. It was installed in 2003 and is intended primarily to protect the Tay salmon run.

3.1.2.1 Design Best Practice

The main design requirements are as for mesh screens, but in this case the bar spacing should be set to prevent the penetration of the fish's head. It should be noted that equation 1 is based on penetration of a square or rectangular opening and may not be accurate for calculating bar spacing. Rectangular section bars or perforated plates are preferable to round-section bars, which are prone to 'gill' fish.

While conventional design practice demands that bar spacing should be calculated in this way for the target fish, smaller fish will not necessarily pass through screens that have spacings exceeding the fish's body width. Travade and Larinier (2001), who investigated a bywash adjacent to a 25-mm-spaced vertical bar screen at St Cricq hydroelectric plant in France, estimated that >90% of salmonid smolts were successfully diverted into the bywash even though they were small enough to pass through the screen. The screen in this case was aligned perpendicular to the flow, which is not recommended in practice, but the width of the channel was small (11m) so that fish readily found the outlet.

Angled bar racks are used quite widely on small hydroelectric plants in north America but Simmons (2000) noted that their performance has rarely been examined. He reported a study of the bypass efficiency of an angled bar rack at the Lower Saranac Hydroelectric Project at Plattsburgh New York, which also used 25 mm spacings. In this case the rack was aligned at 45° to the flow (in plan). In an experiment in which 52 Atlantic salmon smolts were released upstream, 29 passed the Project; none passed through the trash rack. Of 23 steelhead trout (*Oncorhynchus mykiss*) passing the Project in a similar experiment, 3 (13%) passed through the trash racks. These results were obtained under the optimal conditions tested, in which high bywash entrance velocities were maintained ($\geq$1 ms^{-1}, *cf.* velocity perpendicular to the screens of $\leq$0.6 ms^{-1}). Both this and the French study mentioned above emphasise the importance of good hydraulics and bywash attraction flow in achieving high bypass efficiencies with over-spaced bar screens. These aspects are discussed further in Section 4. Where over-spaced racks are used, observational trials will be needed to check efficiency.

Solomon (1991) referred to the possible 'louvre-screen' effect of trash racks placed tangential to the main channel flow; the implication of this is that vortices generated by flow hitting the bars will act as a deterrent to fish. Unfortunately, this is not consistent with this author's (AWHT) observations of flow at tangential trash racks, where the dominant flow at the trash rack face during periods of abstraction tends to be near-parallel to the bars.

Additional points in bar rack design are:

- inclining the screen by 10^o to the vertical facilitates raking;
- the bars need to be sufficiently stiff to maintain the design spacing throughout the screen; this may require horizontal tie-bars to be fixed across the back of the screen;
- manual raking of bar screens is probably only safe and practicable in water depths of <1.0-1.2 m

3.1.2.2 Applications

Vertical bar racks can be considered a suitable alternative for most applications that would otherwise use mesh screens.

3.1.2.3 Fish Species/Life Stages

Vertical bar racks are potentially suitable for screening most fish, subject to the bar spacing being small enough to exclude them. At present the UK user base for this type of screen is quite restricted and therefore there has been little feedback on the relative performance with different species. It could be expected that eels and other anguilliform species would tend to get trapped lengthwise between the bars but this is purely speculative. There is evidence also that eels, when confronted with a bar screen, try to force their way through, rather swimming along the face of them (Richkus, 2001).

3.1.2.4 Ease of retrofitting

Retrofitting issues are similar to those for fixed panel screens.

3.1.3 Rotary Disc Screens

3.1.3.1 Description of Screens

Rotary disc screens were originally designed for the use in sewage treatment works but may also be applied to intake screening. They are based on a series of plastic, stainless steel or high impact glass-reinforced polypropylene discs stacked in a column with spacing between the discs suitable for its specific usage, such as preventing the entrainment of fish or other debris. The discs within each column rotate in the same direction with adjacent columns interleaving. The discs are driven via motors with the direction of rotation matching that of the direction of water flow. Debris and fish will be passed from one column to another until carried away by the flow. There is also a possibility that vibration of the discs may discourage fish to enter the area but this has not been investigated (Turnpenny, 1998).

Within the UK Mono Pumps Limited[5] produce Discreens™ with apertures of 2.5, 5, 9, 13 and 18mm. They have a capacity range of 0-3.7m^3s^{-1} in depth ranges of 200 to 1750 mm. The screens are fitted with comb bars to eject screened solids back into the main flow.

There have been numerous rotary disc screen installations throughout the UK over the past decade with only one or two being used for fish intake screening applications,

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[5] Mono Pumps Ltd. – Martin Street, Audenshaw, Manchester, M34 5JA (0161 339 9000).

www.mono-pumps.com

others being mainly in water and sewage treatment applications. One at Testwood raw water intake, Hampshire (Mr R. Edbury and Mr M. Bridges, Southern Water Services Ltd, personal communication) comprises four disc columns per screen unit. The units are approximately 1 x 1.2m in area and have a handling flow rate of 40MLd^{-1} (0.46 m^{3}s^{-1}) with an approach velocity of 0.35 m.s^{-1}. The screens were designed to prevent the entrainment of smolts and have a gap size of 9mm. The installation was completed in 1997 at a cost of £200k; the installation still consists of the same number of units although one has been rebuilt during this time. The power consumption is fairly low at 4.4 kW resulting in relatively low operating costs. The screens have experienced some problems with weed becoming wrapped around the spindles of the disc, which has resulted in the need for periodic removal and refurbishment of the screens.

An upgrade at the Knapp Mill WTW by Bournemouth and West Hampshire Water resulted in the update of the existing drum screens with six Mono L series Discreens. The Discreens are designed to prevent entrainment of both debris and aquatic wildlife. Four fourteen shaft screens were placed on the lagoon intake and two ten shaft screens on the upper intake. The screens have a handling flow of 0.76 m^{3}s^{-1} and a power consumption of 3.7 and 4.4 kW. The screen is believed to benefit from self-cleaning abilities.

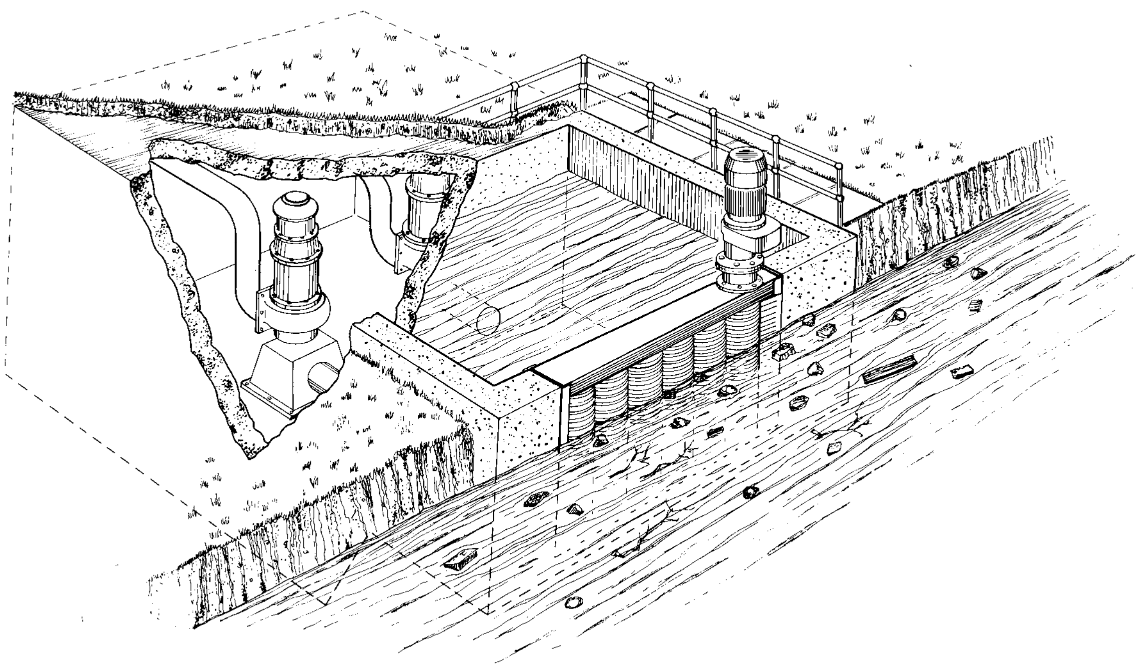

Figure 3.2 Schematic of Rotary Disc Screen

3.1.3.2 Design Best Practice

For any single unit there is a maximum screen depth of 1m. This can be overcome for greater depths via stacking units in a stepped format.

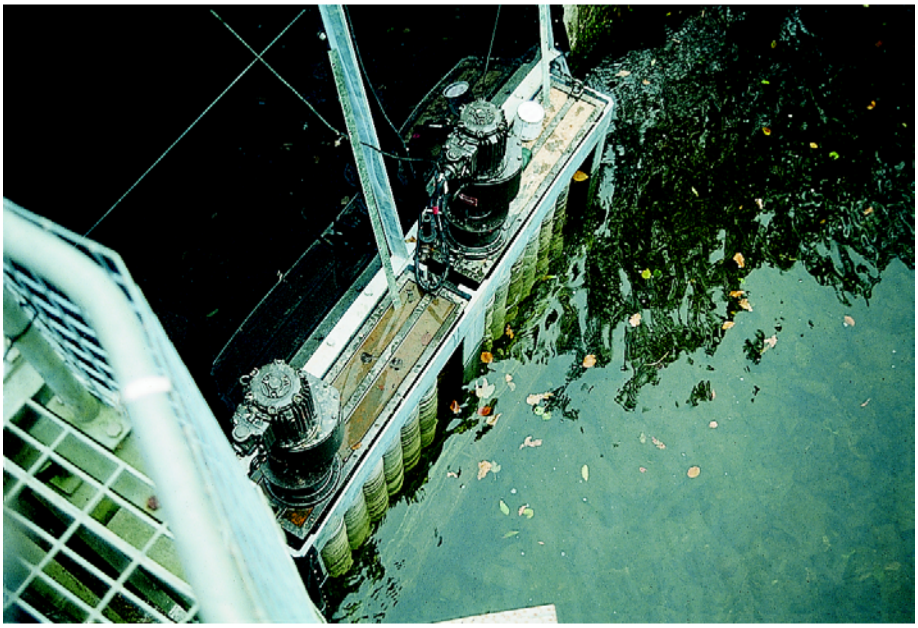

Plate 3.4 Rotary Disk Screen at Testwood Raw Water Intake, R. Test, Hants

3.1.3.3 Applications

The screen is suitable primarily for the screening of smolts and larger fish on rivers with a strong sweeping flow. High leaf and filamentous/stranded weed loads may cause problems. This screen is not economical for large intakes owing to the high surface area required to achieve low approach velocities, and consequent high costs.

3.1.3.4 Fish species/life stages

Depending on the spacing between discs this screen should be suitable for screening salmonid smolts and adults of most species.

3.1.3.5 Ease of Retrofitting

The main attraction of the rotary disc screen is that it is compact and a relatively straightforward retrofit option. It may, as at Testwood, be suitable as a direct replacement for trash racks, provided that these are flush with the riverbank or projecting out into the flow. It can also serve as a replacement for drum screens as at the Knapp Mill WTW. Approach velocities need to be set according to the species and sizes of fish involved.

3.1.4 Spillway Screens

The principle of a spillway screen is that a grid of some sort replaces part of the downstream face of a weir and water falling through the grid enters a channel beneath, from which it is conveyed to the turbine or other application. Meanwhile, fish and debris larger than the screen openings are flushed by surplus flow across the surface of the grid to the downstream side of the weir (Turnpenny, 1998).

3.1.4.1 Coanda Screen

The Coanda screen is based on the 'Coanda-effect', the principle of how fluids follow a surface, a phenomenon first identified by Henri-Marie Coanda in 1910. In this case the surface is that of a wedge-wire screen with the bars running from side to side across the width of the weir (Figure 3.3, Plates 3.4, 3.5). Water then follows the surface of the V-profile wires and runs into the collecting chamber (penstock) below. The wedge-wire screen is contoured to form an ogee-shape curved to a 3m radius. A curved 'acceleration plate' is positioned at the top to stabilize and accelerate the flow. The spacing between the wedge-wire bars is designed to be small enough to exclude all fish including young fry. Depending on the spacing of bars the screen can also be used to exclude silts, sand and gravel (Turnpenny, 1998).

The Coanda screen has been used mainly for small, upland hydro intakes but there is no reason why it should not be used in other applications where the topography is suitable. Within the UK Coanda screens are sold and installed by Dulas Hydro Limited[6] and are manufactured by Optima International[7], Doncaster. The screen is available in a range of designs for varying installation sizes:

Screen A: A full height screen with removable screen material – suitable for flows from 210 $l.s^{-1}$ upward in 70 $l.s^{-1}$ steps.

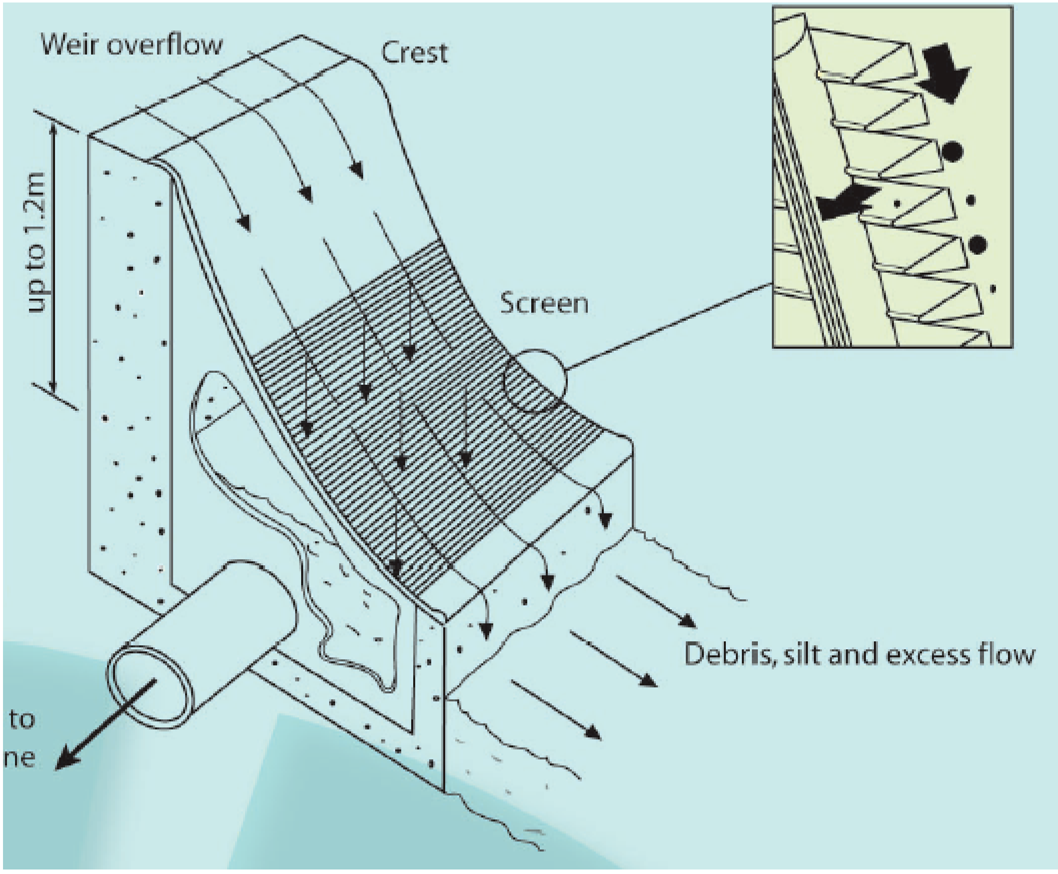

Figure 3.3 Diagram of an Aquashear™ Coanda screen, showing detail of the V-profile of the wedge-wire (Dulas)

[6] Dulas Hydro Ltd. – Dyfi Eco Parc, Machynlleth, Powys, Wales, SY20 8AX.

[7] Optima International – www.optima-international.co.uk.

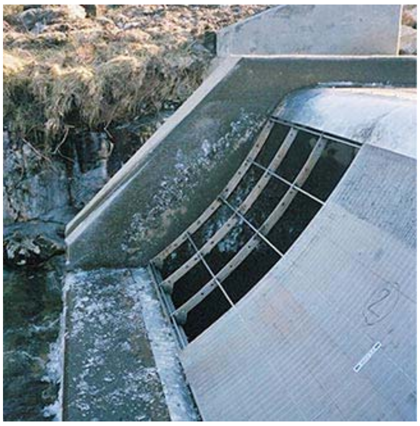

Plate 3.5 Example of a Screen A Coanda installation (Dulas Hydro)

Screen B: A full height one piece small screen – suitable for flows of 80, 100, 120, 140 and 160 ls^{-1}.

Screen C: A half height full width one piece screen – suitable for flows from $100ls^{-1}$ upwards in steps of 50 ls^{-1}.

Screen D: A half height, one-piece small screen:- suitable for flows of 20, 30, 40, 50 and 60 ls^{-1}.

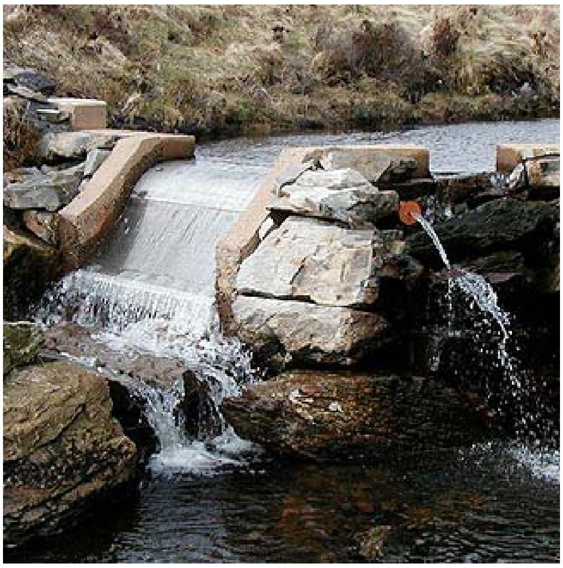

Plate 3.6 Example of a Screen B Coanda installation (Dulas Hydro)

Full height screens have a head loss of 1,270 mm and half height screens have a 705mm head loss. Thus there must be a minimum of 705 mm sacrificable head loss available before a Coanda screen can be used. The maximum flow of the screens is dependent on the weir width. A full-height screen has a capacity of 140 ls^{-1} per metre width, therefore 1 m^3 flow would require a weir just over 7 m in length.

The recommended screen materials are 304 stainless steel in freshwater, or 316 grade for marine environments. The acceleration plate is a circular arc similar to a parabolic ‘ogee’ shape, which matches the path of an unsupported jet of water. The plate acts to speed up the water, helping the shearing effect, which improves abstraction efficiency.

Coanda screens are designed to be low maintenance although during low flows some debris build up may occur, which will be washed off in subsequent high flow periods. Brushing with a stiff broom clears the majority of any remaining debris. Most screens require periodic visual checks and brushing approximately quarterly.

There was a reported total of 22 Coanda screen installations within the UK by 2003. Most are at small-scale private hydropower installations with capacities ranging from 10 to 1,300 ls^{-1}. One of the larger installations was commissioned by Innogy Hydro south west of Fort Augustus in the Scottish Highlands. The screen is a full height design with a flow capacity of 1300 $l.s^{-1}$. Although there were initially some concerns over fouling by algae it has been found that the screen self-cleans during periods of high flow and the overall opinion of the screen at this location is good (personal communication W. Langley).

The effectiveness, suitability and cost benefit of the screen was evaluated at a small hydropower scheme near Keswick in Cumbria (Howarth, 2001). A screen with 1mm bar spacings was commissioned in April 1999. It was capable of excluding all debris greater than 1mm and 90% of particles >0.5mm. Performance evaluation was carried out over a 15 month period with monitoring of screen capacity, silt exclusion performance, self cleaning operation, slime and algae growth, operation and maintenance requirements, integrity and resistance to damage and cost benefit analysis. After the 15-month period there were no noticeable signs of wear and at high flows up to 94% of suspended silt particles between 0.41 and 1.17mm. There had been no records of blockages by debris although it was believed that very thin strands of weed may pass through the wedge-wires. After the 15 months, a thin film of algae had developed over the screen, resulting in some loss of capacity, although it was readily cleaned with a stiff brush. Overall the screen was found to be consistently robust, resistant and has a high performance rating.

Fish Protection Performance

No assessment of fish exclusion efficiency or fish condition after passing the screen was undertaken in the Keswick study. Elsewhere, Bestgen (2000) reported tests carried out at the Colorado State University Larval Fish Laboratory on the exclusion and survival rates of fathead minnow passing over a Coanda screen. Out of 150 trials releasing and recapturing fish downstream of five different lengths (5, 7.5, 12.5, 22.5 and 45mm) an exclusion rate of “nearly 100%” was obtained for fish greater than 12.5mm.

3.1.4.2 The ‘Smolt Safe™’ Screen

The Smolt Safe™ screen (Figure 3.4) is manufactured by Rivertec of East Sussex. The principle is broadly similar to that of the Coanda-effect screen. In the configuration shown, the weir is constructed flush with the bank of the river and water is carried off sideways from below the screen. Water falling through the screen is collected in a take-off channel, while a further debris channel is provided to carry fish and trash back to the river. There is no reason, however, why the screen should not be constructed as part of a transverse weir, as in the Coanda-effect example.

The example shown in Figure 3.4 was constructed at a distillery where there is a large amount of waterborne debris. Screen mesh size in the example is 10 mm, but this can be varied as required. The manufacturers claim the screen to be 100% safe for passage of smolts and other fish but this has not been verified by trials. A similar screen built at Heltondale in Cumbria has been found not to be suitable for screening pre-smolts (G. Armstrong, personal communication. The problem was due to fish becoming trapped among debris at times when there was insufficient washover flow. Potentially, this can be overcome by blanking off part of the screen at low flows but the degree of washover is difficult to control with variable river flows, particularly at remote sites. The fact that the screen is flat rather than inclined (as e.g. in the Coanda screen) does not help debris clearance.

As for the Coanda-effect screen, there are constraints on operation. The manufacturers specify an operating flow range of 0.5 to 5 m^3s^{-1}. However, there seems no reason in principle why larger flows should not be accommodated, given suitable space and arrangement of the civil works. A second constraint is that at least 25% of flow is required for washover. Thus, for a 5 m^3s^{-1} draw-off, at least 6.25 m^3s^{-1} initial river flow would be required.

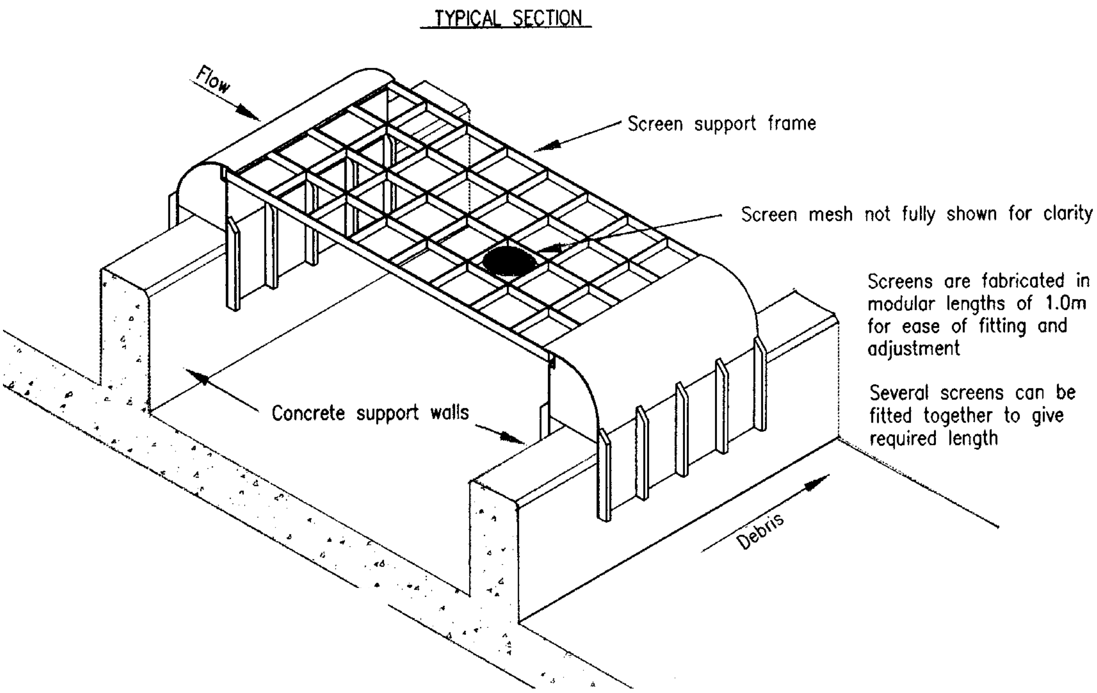

Figure 3.4 The Smolt Safe™ Screen (Rivertec Ltd)

3.1.4.3 Design Best Practice

For spillway screens, the manufacturers recommendations must at present be regarded as best practice. An important consideration is the relationship between abstracted flow rate and the surplus washover flow. If the screen is over-sized, then there may be a risk of not leaving sufficient ecological flow in the river downstream of the offtake and of there not being sufficient surplus flow to flush fish and debris safely off the screen in the downstream direction. It may be necessary to provide a means of blanking off part of the filtration area during dry weather flows. This problem is likely to be greater on flashy upland streams than e.g. in chalk streams having a stable flow regime.

Debris should not be allowed to accumulate on the screen owing to the risk of causing fish injuries.

3.1.4.4 Applications

In the UK, Coanda screens have chiefly been used at medium to high head hydropower screens in upland areas. However, there is no obvious reason why either Coanda or

Smolt Safe screens should not be used for other types of application where sufficient head of water exists, e.g. fish farms located on upland rivers.

3.1.4.5 Fish Species/Life Stages

Although used in this country mainly for trout and salmon exclusion, spillway screens would be equally suitable for exclusion of eels, lampreys and other upland river species. The sizes of fish excluded would depend on the wire spacings but, for example, a ≤3mm spacing should be suitable for salmonid fry exclusion.

3.1.4.6 Ease of Retrofitting

Coanda or Smolt-safe™ screens are likely to be suitable mainly for new-build applications or replacement of existing spillway screens.

3.1.5 Band or Drum Screens Modified for Fish Return

Many power stations intakes and potable water abstractions are fitted with mobile band screens or drum screens for debris filtration. These are usually set somewhere within the pumphouse and not at the intake point. Fish-handling modifications have been developed for both types which can reduce the risk of injury, at least to the more robust species. The modifications relate chiefly to the design of the ledges or 'buckets' which are used to lift fish and debris out of the water, and to reduced-pressure backwash sprays that are used to flush material off the screens and out of the buckets. Thereafter, instead of discharging the filtered material into trash baskets for disposal, a return gully or pipeline puts them back into the water body. Such a system is commonly known as a 'fish return' or 'fish rescue' system.

Fish return systems have been used at power stations in the UK for many years. The earliest ones were constructed at CEGB estuarine sites in the 1960s for the return of salmon *(Salmo salar)* and sea trout (S. *trutta)* smolts (e.g., Uskmouth and Oldbury-on-Severn power stations) but for various reasons these were never fully utilised or evaluated. A number of other stations (e.g. Dungeness 'B', Sizewell 'A') have operated simple trash return systems which involve periodic discharge to the sea of the biological and other debris that has accumulated in trash baskets, with no deliberate attempt to promote the survival of living organisms; in fact, the system at Sizewell 'A' macerates the debris prior to discharge. Sizewell 'B' power station incorporate a facility to direct trash either to baskets or into the CW discharge, along with a number of other engineering measures to reduce stress effects on fish. The Sizewell 'B' system is licensed to operate in fish-return mode, provided that quantities of fish are below a certain level, otherwise fish must be collected in trash baskets to avoid possible wash-up of dead fish (notably sprats) on beaches. Barking Power Station (Thames Estuary) has a fish recovery system based on modified band screens, which returns fish to the estuary via the cooling water outlet.

Fish return is practiced widely overseas. In the United States, the prospect of compensation being levied against fish losses and the requirement under Section 316(b) of the Federal Water Pollution Control Act to implement effective environmental protection systems has encouraged the development of fish return systems. In Europe, the French power industry, as part of its nuclear expansion during the '70s and '80s, investigated a wide range of fish protection systems and has implemented a fish and shrimp recovery system on at least one estuarine site, Le Blayais, on the Gironde (Travade *et al.,* 1983). A new power station at Doel in Belgium is also being fitted with a fish return system, following the demonstration of good fish and shrimp survival rates on the bandscreens (Maes *et al*., 1999).

Plate 3.7 Fish Return System at barking Power Station, Thames Tideway. Insert shows screen panel with fish buckets inverted.

3.1.5.1 Operating Principles of a Fish Return System

The main changes to a standard band or drum screen are to add water-containing scoops or 'fish buckets' at the bottom of each mesh panel, and to use a low-pressure (≤1 bar) backwash spray to flush fish off into the return gullies. A high-pressure spray (≥3 bar) can be deployed at a later point in the cycle to remove the more persistent debris. Rotation speed is also an important factor. Where fish are not a concern, bandscreens are rotated intermittently, either at preset time intervals or when sufficient material has accumulated to cause a head differential across the screen mesh; this save on bearing wear. With such an arrangement, fish may become impinged on the screens for hours before being lifted out by the screens, and may become asphyxiated or exhausted. During rotation, conventional bandscreens operate at one of two speeds, the low-speed setting being used for normal levels of trashing, switching into the faster setting when inundated with trash. To optimise fish survival, the screens are rotated continuously, switching to the higher speed setting if a head loss develops (usually >100 mm) across the screens.

After being washed off the screens by the low-pressure spray jets, the fish and other organisms are flushed into open troughs, and from there to a discharge pipeline that returns them to the water. Handling stress is minimised in these stages through careful design and construction of the gulleys and pipes, ensuring that tight bends are avoided and that smooth surfaces are provided. Swept bends are used throughout with either stainless steel, fibre-glass or PVC materials, with joints ground smooth. The recommended radius for swept bends is 3 m when a trough or pipe diameter of ≤0.3 m is used (Turnpenny *et al*., 1998), although space constraints do not always allow this. Where smaller-radius bends are used, fish tend to find shelter and epibenthic species in particular may accumulate; also, tight bends are susceptible to blockage, hence access for cleaning is required. Where larger pipe or trough diameters are used (≥0.4 m), the bend radius may be reduced to ≥1.5 times the pipe diameter, as there will be less risk of blockage. The chief requirements are that blockage and hold-up of debris should be avoided and that access for maintenance should be provided in case of blockage. The screen wash pumps supply flow for the backwash sprays to ensure that the fish are kept moving through the system and to reduce the risk of blockage.

The use of chlorine or other biocides can, potentially, reduce survival in fish return systems. Fortunately, in the UK, typically 80% of the annual fish take occurs outside the season (approximately May–September, or when water temperature is >9^{o}C) when biocides are applied. The toxicity of chlorine, the most common biocide used at power stations, depends on the concentration and the exposure time. For biofouling control purposes, chlorine is normally injected in the intake at around 1 mgL^{-1}, decaying to about 0.2 mgL^{-1} at the CW pumps. The exposure time in the intake forebay and screenwell can be kept to an hour or less in a purpose-built system. The toxic risk is generally low under these conditions. However, unless a detailed analysis of the toxic risk can be undertaken, taking into account the local water quality conditions, mixing dynamics and species and life-stages exposed, it is preferable to ensure that chlorine is injected downstream of the screens.

3.1.5.2 Fish Protection Performance

The survival rates of the returned fish can depend on a number of design and operational factors. Design variations revolve mainly around the shape and construction material of the fish buckets and the backwash arrangements. Older designs often had incorrect bucket geometry, so that fish fell back into the water and were recycled several times or were not washed out from the optimum point of the cycle. Table 3.1 shows the typical survival rates measured at older fish return systems, ranging from >80% for robust epibenthic species to virtually nil for delicate pelagics.

Table 3.1 Typical fish survival reported from studies of drum-or bandscreens with simple modifications for fish return (e.g. with fish buckets, low-pressure sprays and continuous screen rotation) (Turnpenny, unpublished data).

Fish Group	Survival Rate >48 h After Impingement
PELAGIC	
e.g. herring, sprat, smelt	<10%
DEMERSAL	
e.g. cod, whiting, gurnards, etc	50-80%
EPIBENTHIC	
e.g. flatfish, gobies, rocklings, dragonets, etc., and crustacea	>80%

In the USA, requirements to reduce fish impingement mortalities have led to renewed research into fish return techniques. Recent developments have benefited in particular from the use of CFD flow analysis to optimize the fish bucket design, which can greatly improve the fish retention and reduce damaging turbulence. There have also been improvements resulting from use of non-metallic buckets, smoother screening materials and improved methods of washing off the fish. One company (Beaudray USA) claims to improve fish survival by removing fish before the screen lifts them out of the water. It is now claimed to be possible to return >90% of even delicate pelagic species alive.

3.1.5.3 Design Best Practice

With recent developments in the USA, this is clearly an area where improved designs shortly may become more widely available. Some of the innovations are likely to be protected by patents and therefore available only through certain screen manufacturers. It is important when specifying band or drum screens which are to be used for fish return to ensure that the design of the fish buckets in particular has been optimized for fish handling and evidence of this should be sought from the manufacturer. Other key points in fish return system design are:

- The screens should be capable of long-term continuous operation: intermittent operation is unsuitable for fish return. This means, in particular, that bearing life should be considered.
- The screen meshes should be smooth and ‘fish-friendly’. Certain types of woven stainless mesh are commonly used for this purpose.
- The mesh size should be as small as is practical, and of no more than 6 mm aperture.
- Low-pressure backwash sprays (≤1bar) should be used for fish removal; higher pressure jets may be used at a later point in the cycle to wash off debris.

- The geometry of the collecting hoppers should be checked to ensure that fish that are washed off the screens cannot fall back into the screenwell (an issue mainly on drum screens).
- Biocides should be applied downstream of the screens unless it can be shown that the toxic risk is negligible.
- Fish return gullies should be smooth, with any joints properly grouted and finished. They should be a minimum of 0.3m diameter; 0.5m diameter or larger is preferred for long runs (>30m).
- It is beneficial to enclose or cover fish return lines to avoid algal growth. Suitable access hatches or rodding points should be provided to facilitate maintenance.
- Where bends are required, swept bends of radius >3m should be used.
- Dedicated fish return lines which discharge well blow the low water mark are preferred. Return on power plants via the heated water discharge should only be used where it can be demonstrated that survival rates will be acceptable.
- A continuous washwater supply should be provided that will ensure sufficient depth to keep fish immersed and moving along the return line.
- At coastal sites where there is a risk of occasional inundation by schools of pelagic fish, provision may need to be made for diverting the catch to collecting baskets. This can be necessary to avoid the risk of discharging large quantities of dead fish onto neighbouring bathing beaches.

Plate 3.8 A sharp bend in this fish return gulley demonstrates the issue of biofouling where uneven joints have encouraged algal growth, causing a flatfish to take cover there. A smooth surface or exclusion of light would have prevented this.

3.1.5.4 Applications

Fish return systems are presently used mainly at estuarine and coastal power stations, although the technique is potentially suitable for fish protection at potable water intakes where band screens are often used.

3.1.5.5 Fish Species/Life Stage

In the past, fish return systems have been suitable mainly for more robust epibenthic species, such as flatfishes and reef/rock-pool species, with moderately good results for demersal fishes such as cod and whiting but with very poor survival prospects for delicate pelagics (Table 3.1). With improved designs, some systems may also be suitable for pelagic species. Fry are normally too delicate to survive handling in this type of system.

3.1.5.6 Ease of Retrofitting

In most cases, a system designed without fish return facilities will require substantial modification of civil works to accommodate larger buckets, as well as fish return ways. Careful analysis of the system by a specialist in this field may suggest modifications that would substantially improve fish protection, however.

3.1.6 Econoscreen

The ‘Econoscreen’, a self-powered rotating drum screen as described in Solomon (1992), appears to be unavailable at the time of writing. This is unfortunate, as results have appeared promising at the few sites where it has been used. They include Shotton steelworks (R. Dee) and an abstraction in Port Talbot (D. Mee, Environment Agency, personal communication).

3.2 Physical Screening for Juveniles and Small Fish

Of the screening techniques described above, other than the Coanda screen the methods are generally unsuitable for screening juveniles and alternatives should be considered. The methods described below are, of course, highly effective against larger fish as well, but (mainly on cost grounds) would not generally be used where it was not also necessary to screen out small fish. These methods, with appropriate design, can be used to screen fish even down to larval or egg size.

3.2.1 Passive Wedge-Wire Cylinder Screens

Passive wedge-wire cylinder (PWWC) screens are a tried and tested solution and are generally regarded in Britain as the best available technology for juvenile and larval fish protection.

3.2.1.1 Basic Form of the Screen

Figure 3.5 illustrates the basic form of the PWWC screen. It comprises a cylinder, formed of the wedge-wire material around its circumference, one end being blanked off and flow being drawn off through the opposite end. The blanked end may be closed off either with a flat plate or, where facing into a flow, with a conical cap for streamlining.

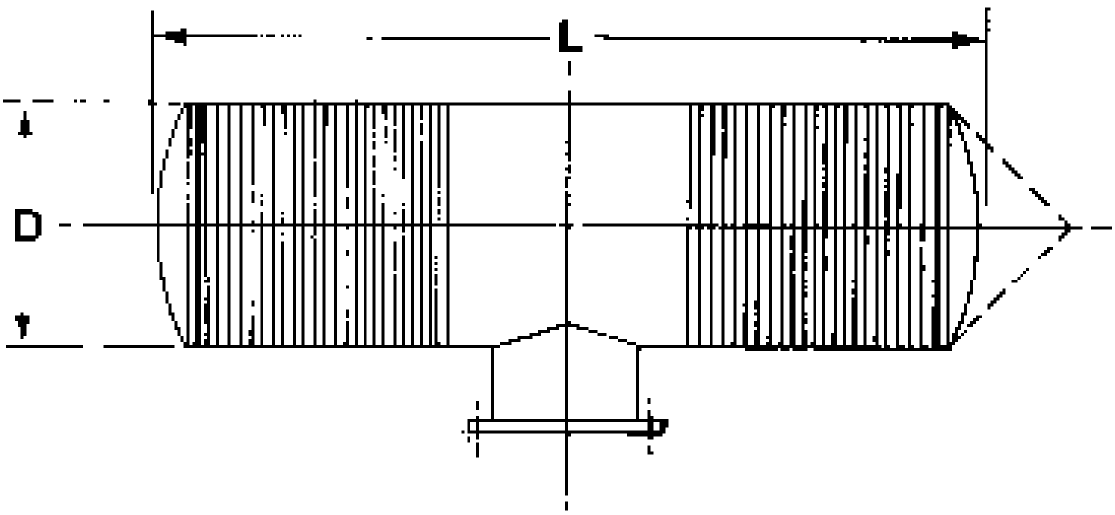

Figure 3.5 The basic form of the Passive Wedge-Wire Cylinder (PWWC) Tee-Screen (courtesy Johnson Screens)

The wedge-wire material is similar to that used in the Coanda screen (Figure 3.3). The profile of the wire that forms the screen surface is V-shaped. In manufacture, the longitudinal supporting bars are fixed about a mandrel, around which the vee-wire is wound in a spiral. The apex of the 'V' is welded onto the bars at each point of contact. The pitch of the spiral thus determines the slot-width of the screen. For some applications, the wedge-wire screening material is deployed in the form of flat panels, but it will first have been manufactured by this method and then flattened out.

The major benefits of using the V-profile wire in PWWC screens are that it offers low hydraulic resistance for a given open area (when compared with conventional screening materials), combined with low blocking risk: particles tend either to wash past the screen or to pass through the slots, as slot width increases towards the inside of the screen.

3.2.1.2 PWWC Screen Configurations

Manufacturers offer a range of PWWC configuration options, including single, bulkhead or pipe mounted units, tee-form screens and multiple groupings attached to a manifold. Figure 3.6 illustrates various typical arrangements. The arrangement used depends on the water depth, space available and other factors, but the options available make the configuration very flexible. Where, for example, water is shallow, a number of small-diameter units can be used rather than a single large one.

3.2.1.3 Air Backwash System

Although not fitted to all systems, PWWC screens are more often than not fitted with an air-blast backwash system, such as the Johnson Hydroburst™ system. In this, a perforated air discharge pipe is welded along the bottom, inside of the screen. This is fed by an air compressor and reservoir, from which explosive bursts of air (up to 10 bar pressure) are released at regular intervals (e.g. daily or more often, depending on debris levels), or else once a certain pressure differential has been measured across the inside and outside of the screen. This may be under manual or automatic control. The clearing action is caused by the displacement of water through the slots from inside the screen chamber, as the air volume expands following release. Any debris that has become pinned on the outer surface is thus lifted off and carried away by local water movement.

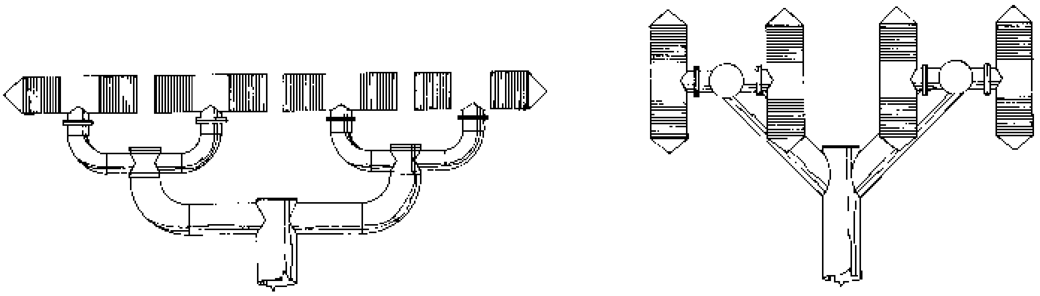

Figure 3.6 Examples of multiple PWWC screen arrangements. These usually involve connection to a manifold (courtesy Johnson Screens).

3.2.1.4 Construction Material and Biofouling

For freshwater use, screens are made from stainless steel, of a grade suited to water quality conditions. In marine and estuarine environments, stainless steel screens tend to biofoul rapidly and a copper-nickel alloy is preferred. Bamber and Turnpenny (1986) tested the efficacy of a small 70%:30% Cu:Ni test PWWC screen at Fawley Power Station on Southampton Water, Hampshire where the mean salinity is around 32‰. It showed little sign of biofouling after 15 months of operation without any cleaning other than the once-daily air backwash cycle. After this time the measured flow throughput was reduced by only 2% compared with the starting figure (nominal flow rate $10Ls^{-1}$). More recently, an alloy of 90%:10% Cu:Ni composition has been used for estuarine applications. A large cooling water make-up intake at Connah's Quay Power Station on the Dee Estuary (Cheshire) with PWWC screens constructed of this alloy has operated continuously since 1996 without any need for cleaning (W. Smith, PowerGen plc, personal communication).

3.2.1.5 Fish Protection Performance

PWWC screens have a number of features that make them suitable for prevention of fish entrainment. These include the low through-slot velocity, allowing fish to swim away, the relatively smooth external presentation of the screen, which reduces the risk of fish abrasion, and the narrow slot widths available, making it possible to prevent entrainment of fish even down to egg or larval sizes. The main reason for selecting PWWC screens in preference to lower cost alternatives is to improve the level of protection for the smallest individuals, i.e. egg or larval/postlarval stages ('pinhead fry'). This aspect has been investigated in North American studies (Heuer and Tomljanovich, 1979; Hanson, 1979).

Conclusions of the Heuer and Tomljanovich (1979) study were:

- For very small larvae (<6.0 mm total length), a slot width of 0.5 mm and through-slot velocity of ≤ 7.5 cm.s^{-1} would be required.
- For larvae of 7-10 mm total length, a slot size of 1.0 mm and through-slot velocity of 7.5 cm.s^{-1} was ideal, although a through-slot velocity of 15 cm.s^{-1} would be low enough for some species.

- For larvae of >10 mm total length, a slot width of ≥2.0 mm is satisfactory, with a through-slot velocity 7.5-15 $cm.s^{-1}$.

While close to 100% larval exclusion was achieved with the lower through-slot velocity and a slot size of ≤1.0 mm, significant entrainment of some species occurred at 15 $cm.s^{-1}$ through-slot velocity and 2.0 mm slot width, e.g. 18.1% for bluegill (*Lepomis machrochirus*) and 67.7% for channel catfish (*Ictalurus punctatus*).

Entrainment of fish eggs and larvae was also studied by Hanson (1979), in a laboratory flume with 1 and 2 mm slot-widths and a 15 cms^{-1} through-slot velocity. The particular significance of this study was that they measured the effect of the channel velocity (0.15, 0.3 and 0.6 $m.s^{-1}$) on fish entrainment rates, after releasing batches of fish eggs and larvae into the flume. The findings are summarised in Figure 3.7, which expresses the results as the percentage of fish exposed to the screen that became entrained. This indicates the importance of placing screens in a strong flow (>0.3ms^{-1}) if the best performance is to be achieved.

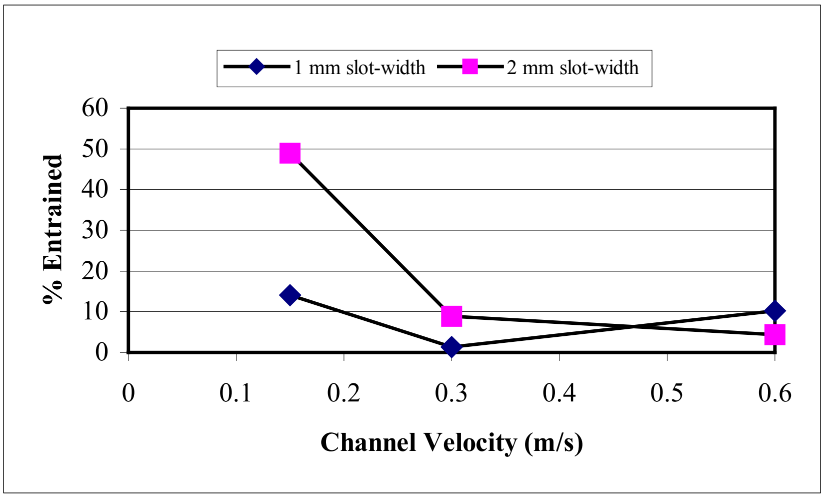

Figure 3.7 Entrainment rates of fish eggs through 1 mm and 2 mm slot-width screens at different channel velocities (after Hanson, 1979).

3.2.1.6 Design Best Practice

Manufacturers offer design guides that provide the information required for specifying the screening system. A detailed account is not therefore necessary here but the main points will be listed. The following information is taken largely from the Johnson Screens' guide.

3.2.1.7 Through-Slot Velocity.

The design velocity is commonly 15 $cm.s^{-1}$, a value that has been found to offer virtually maintenance-free performance of the screen. As screens seldom are operating in the fully clear state, a degree of occlusion is allowed for when sizing the screens. An allowance of 25% is normally made.

3.2.1.8 Slot Width.

Typical values used range from 0.5 mm to 9.5 mm. The very small slot widths may be used, e.g., where there is a risk of sand ingress. The most common size used in the UK for raw water screening is 3 mm, this being a reasonable compromise between open area and effective debris screening. Also, the smaller the slot width used, the larger the overall screening area required and the higher the capital cost and space occupied.

3.2.1.9 Screen Diameter and Spacing from Surfaces

The maximum screen diameter should be half the water depth at the lowest extreme of water level; preferably it should be no more than one-third. Where depth is shallow, the option of using tee-configurations or other multiple arrangements of small-diameter screens can be considered.

The recommended minimum submergence depth is half the screen diameter, with the screen being spaced an equivalent distance from the bed and any wall. Submergence to this depth avoids the risk of excessive entrainment of surface-carried debris into the abstraction flow. Spacings from the bed and wall are to avoid debris rolling along the bed becoming entrained, or larger items becoming jammed. Placing screens too close to the bed or wall may also compromise the uniformity of the hydraulic field around the screen.

3.2.1.10 Screen Sizing

The number, types and sizes of screen units required for a given abstraction are selected so as to satisfy the above requirements. Manufacturers' design guides provide tables and formulae from which requirements can readily be calculated.

3.2.1.11 Velocity of Flow Past Screen and Screen Siting

The successful clearance of debris following air backwashing is dependent on adequate ambient flow past the screen, otherwise, debris may accumulate. This may be through river or tidal flow, or through wind-driven circulation in lakes and reservoirs. It is also important that screens are not sited in backwaters where debris naturally accumulates as a result of eddy currents.

A steady current is required to ensure debris is carried away.

3.2.1.12 Applications

PWWC screens are suited to a wide range of flowing water applications in freshwater, estuarine and marine environments. They are best suited to smaller abstractions of a few m^3s^{-1} or less, as larger arrays may become cumbersome, unless space is unlimited. They are used, for example, for potable water abstractions, CCGT[8] power stations and fish farm supply, but are not suitable e.g. for low-head hydroelectric generation on account of the very large flows involved.

3.2.1.13 Fish Species/Lifestages

They are probably suitable for excluding all species and sizes of fish given suitable wire spacings. A particularly interesting case is the study carried out by the National Rivers Authority at Moor Monkton pumping station on the Yorkshire Ouse (Frear and Axford, 1991 and unpublished). Collections of impinged fish from the bandscreens were made before and after the fitting of PWWC screens to the intake. Between January 1990 and May 1991, 16,022 lampreys (brook- and river-) were collected from the band screens; most were recently metamorphosed pre-adults ("transformers"), along with some ammocoetes and adults. In 1995, the intakes were fitted with an array of eight Johnson PWWC screens (model T42 with 3 mm slot-width, total capacity 3.5 m^3s^{-1}). Subsequent surveys found virtually no lamprey or other fish impingement. The small numbers that were collected (around ten per week during the winter) may have passed through the screen but could also have been ones that remained resident in the abstraction lagoon,

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[8] Combined-cycle gas turbine

between the intake and the band screens. Samples of lampreys retained in tanks following impingement indicated potentially high survival rates, suggesting that returning lampreys from the bandscreens to the river via a fish return system would be an option worth considering.

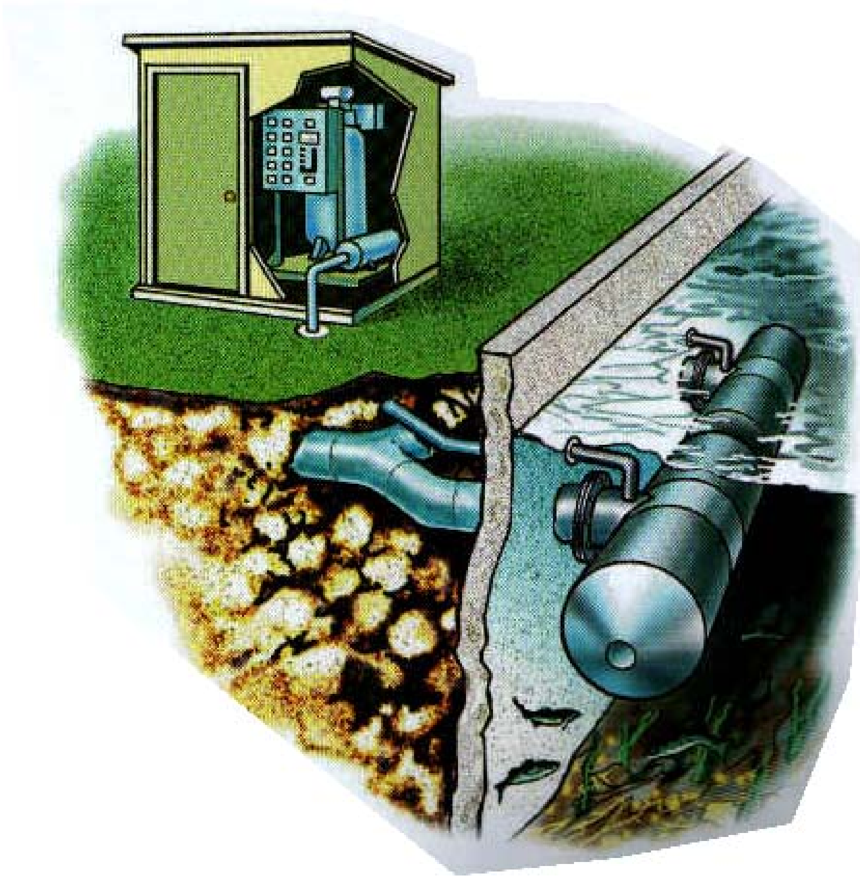

Figure 3.8 Illustration of bulkhead mounted PWWC screen array with manifold and air backwash system; the compressor is located in a bankside hut but could be located in any neighhbouring building. (courtesy Johnson Screens)

3.2.1.14 Ease of Retrofitting

PWWC screens do not lend themselves to retrofitting on existing intakes, except perhaps for end-of-pipe applications. Where the existing intake is constructed e.g. as an open channel or opening in the riverbank protected by a trashrack, it would be necessary to form a bulkhead onto which a screen manifold could be fixed (Figure 3.8). However, PWWC screens do have the advantage that they are available in a wide range of dimensions, so that many different configurations can be achieved.

3.2.2 Wedge-Wire Panel Screens

Wedge-wire can be used in flat panel screens (see section 3.1) as an alternative to mesh panels. This is found to be more practical for small fish exclusion, being less prone to clogging, than a mesh of equivalent spacings. An example of this type of screen was installed at a small hydroelectric plant in the Thames catchment at Huntsmoor on R. Colne (G. Armstrong, personal communication).

North American experience is that orienting the wires vertically rather than horizontally facilitates cleaning, as vertical raking machines can be adapted for this purpose (S. Rainey, US National Marine Fisheries Service, personal communication). However, use of the material in the PWWC screen format is to be greatly preferred, as the air-backwash system provides a very effective cleaning mechanism; air backwashing cannot be used in a vertical flat screen layout, as the air needs to rise through the gaps between the wires.

3.2.3 Sub-Gravel Intakes and Wells

Solomon (1992) discussed the applications of sub-gravel intakes and wells, since when there has been no real change in the approach. A brief summary of these technologies and examples is included here for completeness.

Sub-gravel intakes use the riverbed itself as a screen by abstracting the water from underneath the bed or from an aquifer. This system also has the advantage of natural filtration reducing treatment costs but has the drawback of there being an extremely limited number of suitable locations.

An example of this form of abstraction is found at Ibsley on the Hampshire Avon. An abstraction of 0.57 m^3s^{-1} is taken via 4 streams. A wedgewire screen with 8mm slot width is supported over a concrete chamber over which layers of gravel are placed up to the original bed level. A geomembrane sheet is placed between gravel layers and gravel cleanliness is maintained by backwashing.

Littlehampton abstraction uses a 4 m diameter collector well reaching down to bedrock at 10m below the riverbed. 12 lateral perforated pipes extend from the well at 2 depths. The abstraction licence granted to the well is for 0.28 m^3s^{-1} and is anticipated to benefit from both high fish protection and partial water treatment (Solomon, 1992).

3.2.3.1 Applications

This technique is only feasible for small abstractions in fast-flowing, eroding-substrate rivers and is suitable e.g. for potable water or fish-farm supply.

3.2.3.2 Fish Species/ Life Stages

The technique should prevent the entrainment of any fish present. It may be conjectured that the requirement to backflush periodically could affect the habitats of lithophilous fish/lifestages, e.g. bullheads, stone loaches or juvenile salmonids, but probably over an insignificant area.

3.2.3.3 Ease of Retrofitting

This is unlikely to be a method suitable for retrofitting in many circumstances.

3.2.4 Microfiltration Barriers

The Marine Life Exclusion System (MLES™), developed in the USA and patented by Gunderboom Inc.[9] is a new microfiltration barrier that is presently being tested widely in the USA for fish exclusion at power plant intakes. It is specifically intended to provide

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[9] Gunderboom, Inc., 10 Hickman Dr., Sanford, Florida 32771 USA

protection for early life stages of fish. In the USA, the MLES is a contender for Best Technology Available status under the Clean Water Act, Section 316(b).

The following information is taken mainly from the company's website (www.gunderboom.com) and from correspondence with the manufacturer. The MLES was also reported and discussed in a number of papers presented at the US Environmental Protection Agency's Cooling Water Symposium held in Washington DC in May 2003: presentations may be viewed at the following URL: www.epa.gov/waterscience/316b/symposium. [10, 11, 12].

3.2.4.1 Description of the MLES Barrier

Gunderboom's MLES™ is a water-permeable barrier (Plate 3.9, Figure 3.9) that keeps fish eggs, larvae and other aquatic organisms away from the water intake. Comprised of a pocket formed by two layers of treated geotextile fabric, the curtain is arranged to full water depth across the front of the intake. It is made long enough to provide a very large surface filtration area, with typical velocities through the fabric of only 4-10 mm s^{-1}. The curtain is either suspended by flotation billets and anchored in place, or integrated into existing shoreline intake structures. The curtain fabric is porous, with pore sizes of <1 mm.

[10] Development of Filter Fabric Technology to Reduce Aquatic Impacts at Water Intake Structures, Matthew J. Raffenberg, Lawler, Matusky and Skelly Engineers, LLP

[11] Vulnerability of Biofouling of Filter Curtain Materials Used for Entrainment Reduction, Peter Henderson, Pisces Conservation Ltd. & University of Oxford and Richard Seaby, Pisces Conservation, Ltd

[12] Effectiveness, Operation and Maintenance, and Costs of a Barrier Net System for Impingement Reduction at the Chalk Point Generating Station, David Bailey, Mirant Mid-Atlantic.

Plate 3.9 Gunderboom MLES barrier in place around an intake structure. The yellow support collar is visible and air backwashing is taking place along part of the barrier.

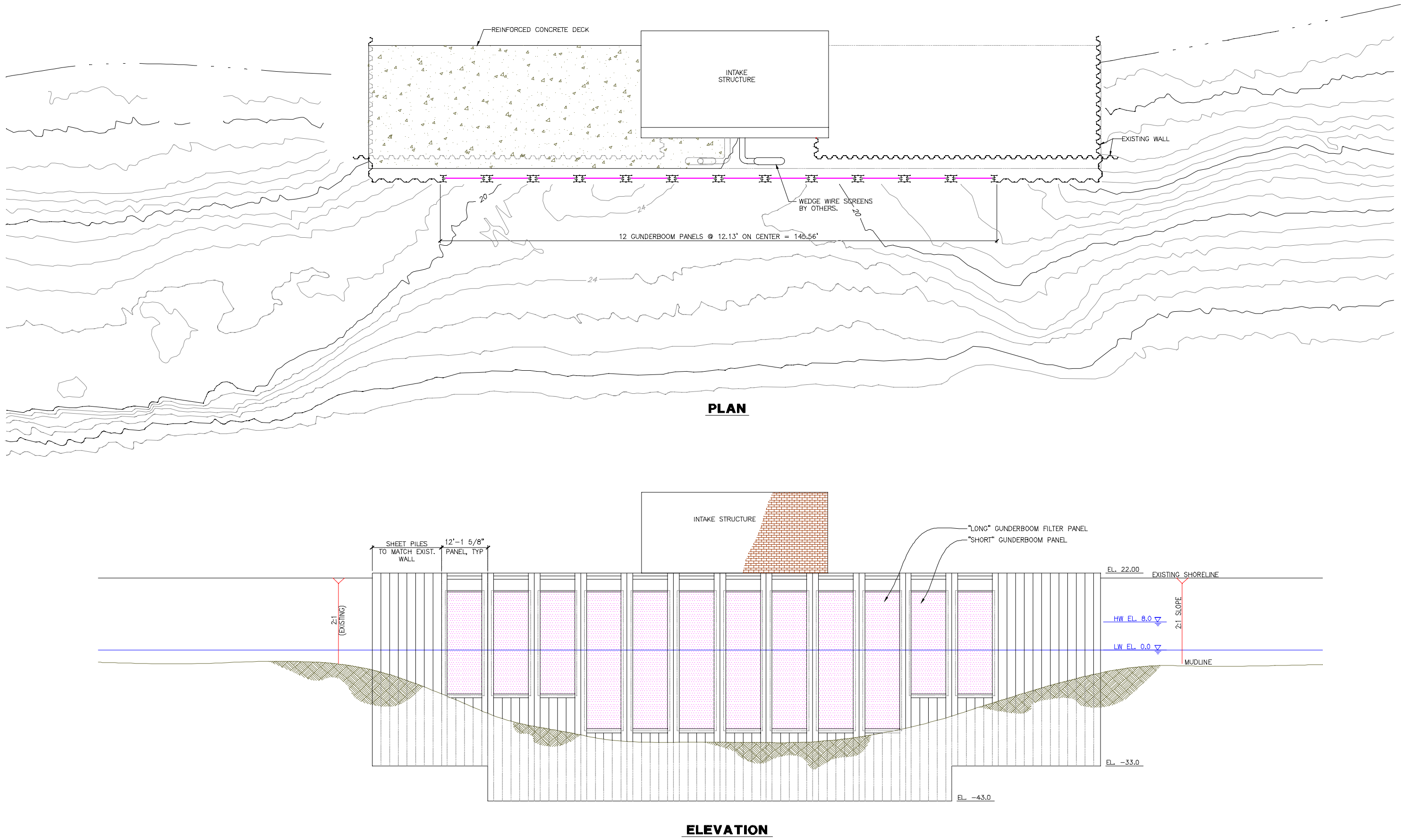

Figure 3.9 Example of a MLES layout (Gunderboom Inc.)

3.2.4.2 Self-Cleaning with AirBurst™ System

Similar to the PWWC screens, the MLES barrier uses an automatic AirBurst™ cleaning system. This is intended to remove sediment and organisms that are drawn onto the fabric when high-pressure air (8 bar) is released at the base of the curtain. Bursts of compressed air shake the fabric panel.

Presumably some tangential flow would be required to carry away debris but the manufacturers do not mention this.

3.2.4.3 Fish Exclusion Performance

As the pore size of the fabric is small, the MLES will *potentially* prevent the passage of all fish down to egg and larval size. Its effectiveness in practice will depend on the integrity of the curtain (freedom from tears, split seams, etc.), achievement of a good seal on the riverbed and banks or intake walls and the ability to resist overtopping by wave action. Raffenberg *et al.*[3] estimated an 80% reduction in larval fish entrainment at the Lovett generating station (USA) and while this might be bettered in some environments, it is unlikely that breaches by one of the above mechanisms can be entirely avoided.

A further concern is the development of a biofouling community. Despite manufacturer's claims that the MLES fabric is biofouling resistant, Henderson[4] monitored biofouling development over a 30 day period and showed that a diverse community can rapidly develop. His concern was not just over the potential for blinding of the pores with consequent lack of flow but also over the arrival of a number of known predators of larval fish, including ostracods, amphipods and crabs. He drew attention to the possibility that these organisms might crop some of the fish larvae, reducing the benefit of the MLES technology.

3.2.4.4 Design and Best Practice

Typically, the MLES technology is used for industrial and power plant applications where the through-fabric flow rates are in the range of 4-10 $Ls^{-1}m^{-2}$ (although the manufacturers suggest that through a combination of modification to the fabric and alterations to the perforation parameters, it is possible to reach higher sustainable flow rates). Barriers are designed to operate at a maximum of 50 mm head differential.

Using the AirBurst™ cleaning technology in conjunction with what the manufacturers claim to be a relatively non-biofouling fabric, the filtration-curtain design flow should be maintained. At present there is some skepticism in North American power plant circles about the generality of this claim and it would be unwise to invest heavily in MLES systems without undertaking pilot-scale site trials to prove the point. In the event of the curtain becoming temporarily blinded by debris, the system can be designed with relief mechanisms such that the operation can return to normal after the adjustment. This is accomplished by having the flotation sized to overtop at certain head differentials and by sizing the ballast on the bottom of the curtain to lift off the bottom given certain predetermined loading parameters. Large concrete anchors (e.g. 3m x 2.4m x 1.8m) are generally required.

At Chalk Point generating station, Bailey[5] reported the need to remove the barrier periodically for cleaning and repairs and allowed for 25% replacement of MLES fabric panels per year. This would need to be done at a time of the year when entrainment risk was low.

3.2.4.5 Applications

The uses of this technology are less obvious in a UK context than in North America, given the generally smaller sizes of water bodies, other than e.g. at estuarine and marine power plant sites. Biofouling, in any case, would almost certainly preclude the use of this approach in saline waters. At present, it cannot be recommended as an ‘off-the-shelf’ solution for UK waters. Nevertheless, it may be amenable for use at lake and reservoir offtakes, and perhaps in slow-moving lowland rivers and canals where space allowed without jeopardising navigation. Suitable trialling of the system would first be required.

3.2.4.6 Fish Species/Life Stages

Suitable for exclusion of all fish, down to egg and larval size.

3.2.4.7 Ease of Retrofitting

The MLES™ is intended as a retrofit ‘fix’ for existing abstraction plants and, because of its simplicity, is likely to be an easy retrofit, provided that space and environmental conditions (wave climate, boat traffic etc.) are suited.

3.3 Other Positive exclusion Fish Screens

A number of other positive exclusion fish screening methods are used or being trialled overseas, especially in North America, none of which have so far been introduced into the UK. In some cases this may simply be a matter of the larger scale of North American facilities and waterways but it is likely that we can learn from these techniques and adapt them for UK use. It would be premature to present them as “best practice” at this stage. Some of the material presented here has not been formally reported in publications. A number of the newer ideas were presented at a recent meeting on intake screening technologies organised by the Electric Power Research Institute (EPRI) at the Alden Laboratories in Massachusetts, USA (30 September 2004). Copies of the presentations are due to be released on the Internet by EPRI (epri.com).

3.3.1 Barrier Nets

Fish barrier nets have been used at a number of large US power stations to reduce fish impingement on cooling water screens. These are large nets that are arranged in an arc in front of the intake and can be several kilometers in length. They are therefore mainly suited to large water bodies. The size of the mesh needed is a function of the species present, typically varying from 4 mm to 32 mm. There is a risk of gilling fish if the meshes are too large. Diamond meshes are preferable to square meshes, as they do not deform so easily. Design approach velocities are kept to $\leq 7.5 cms^{-1}$.

The nets are supported on piles spaced 3-12m apart and may be deployed from shore-mounted drum winches, allowing retrieval for maintenance and cleaning. They may be arranged in two tiers, so that a clean net can be put in place before the soiled net is removed. Maintenance requirements depend on debris and biofouling levels, but would typically be every few weeks. Excessive fouling can cause the nets to lift from the bottom.

Barrier nets are most suitable for environments with low biofouling and debris levels, and where the fish risk is seasonal, so that they do not need to be in place all year round (as e.g. for many smolt screen installations in the UK).

3.3.2 Modular Inclined Screen

The Modular Inclined Screen (MIS: Figure 3.10) is a new type of fish screen from the USA, designed by the Electric Power Research Institute (EPRI)[13] to suit a variety of different water intakes, fish species and sizes (Amaral *et al.*, 1999).

The screen is formed from wedge-wire and is angled at 10-20° (relative to the horizontal) to the flow. The wires are spaced at approximately 1.9 mm to give 50% porosity. The screen is placed on a pivot to aid in rotation for cleaning via backflushing. A bypass system is provided for guiding fish to a diversion channel. A full-scale model of the screen will be approximately 9 m in length and 3 m in width. The system is completely enclosed and has a capacity of $2.8 m^3 s^{-1}$ at $3 m.s^{-1}$. It is designed to operate at a velocity of $0.6\text{-}3.0 m.s^{-1}$.

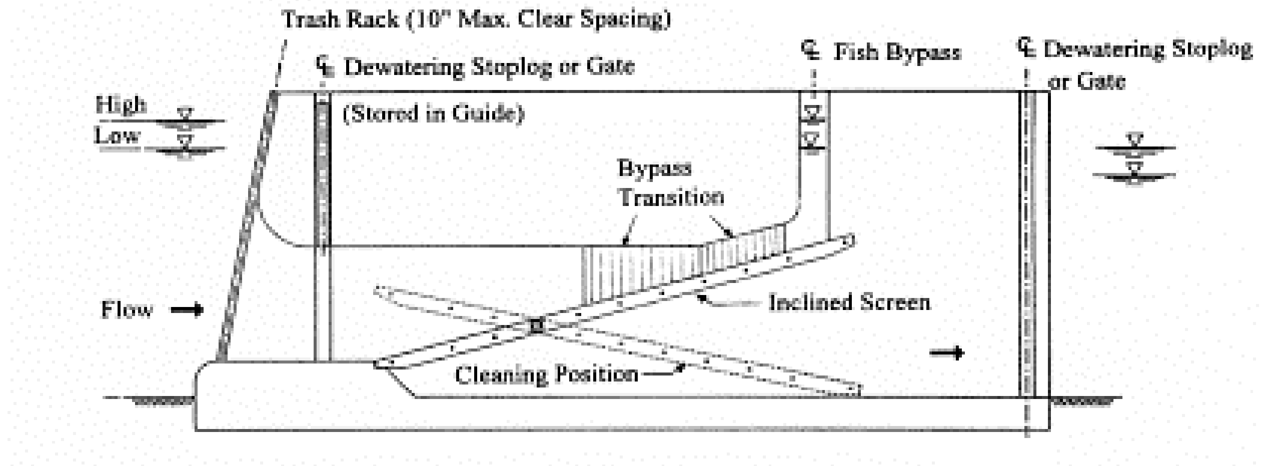

Figure 3.10 Diagram of a Modular Inclined Screen (www.aldenlab.com/scop-fisheries)[14]

Amaral *et al.* (1999) describe laboratory tests carried out in 1992 and 1993 to determine the efficiency of the system. The fish species evaluated included a variety of salmonid and clupeid species such as coho salmon (*Oncorhynchus kisutch*) and rainbow trout (*Oncorhynchus mykiss*). Diversion effectiveness was evaluated for a series of different approach velocities from 0.6 to $3.0 ms^{-1}$. The percentage of live fish that were diverted exceeded 96% for all velocities. In particular Atlantic salmon smolts were diverted with a 100% survival rate for all test velocities.

The success of this laboratory investigation led to a prototype being investigated in the field. The prototype was installed at the Green Island Hydroelectric Project, Hudson River, New York in 1995 and 1996. The facility had a trashrack at the entrance of the MIS and a transition wall guiding fish to a bywash entrance. Tests were conducted at velocities of 0.6 to $2.4ms^{-1}$. The passage survival and live diversion rates exceeded 95% for many riverine species tested (Amaral *et al.*, 1999).

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[13] Electric Power Research Institute (EPRI), 3412 Hillview Avenue, Palo Alto, California, 94304, USA.

[14] Alden Research Laboratory, 30 Shrewsbury street, Holden, MA 01520-1845, USA.

In principle this would appear to offer a good solution to protecting juvenile fish such as elvers, lampreys and coarse fish at run-of-river hydroelectric projects but large size and high costs relative to flow may in practice limit application to higher head sites, where Coanda screens already have a track-record.

3.3.3 Self-Cleaning Belt Screens

This concept was presented at the EPRI meeting of 30/09/04 (Mr. Greg Gerow, FPI, personal communication) as a possible method for power plant cooling water screening. The screening system is similar to a band-screen, comprising a continuously moving conveyor belt of fine mesh (2.4mm) but whereas band screens are normally used within the plant, some way downstream of the intake, the is screen is fitted at the primary intake point on the river or water body. The screen described was operated inclined at an angle of 54^{o} to the horizontal, below the water at the lower end to screen the water intake; the upper part emerged through the water surface to deposit accumulated debris. The longest practical screen length is 15m. The screens can be deployed side-by-side to increase the filtration area.

The mesh is a ‘fish-friendly’, smooth stainless woven material. The screens have been used widely for screening irrigation intakes, with >700 systems installed. There has been no reported evidence of fish loss at existing installations although formal testing does not appear to have been carried out. A large surface area and low approach velocity (0.15ms^{-1}) are used with the aim of not impinging fish at all.

Plate 3.10 Example of a self-cleaning belt screen installation. The screens are sealed at the sides to prevent fish or debris entrainment; screened weed and other debris are dumped on the ground below the top of the screen (courtesy of FPI Water Screens, USA: www.fpi-co.com)

This type of screen appears suitable for a wide range of applications where self-cleaning, fine-meshed screens are required and may provide a more cost-effective alternative to PWWC screens in some cases. Being of stainless steel construction, it will not be proof against biofouling and therefore it is likely to be suitable only for freshwater applications. At exposed sites, trash racks may be required upstream to protect the fine meshes from flood damage. The design is well suited to intakes that lie flush with the bank and it may offer a retrofit option for many bankside intakes that are presently protected trashracks alone.

3.3.4 Labyrinth Screens

Labyrinth screens are a variation on the flat panel screen or bar rack described in section 3.1. In this case, vertical bar racks are arranged in chevron-formations (when seen in plan view: Figure 3.11), rather like an array of fyke-nets. The fish are guided into bywashes located at the downstream angle of the 'V'. The bar spacing can be specified as usual, according to the sizes of fish to be excluded.

Meritec[15], source of the following information, recently reviewed the labyrinth screen for possible application at a large water intake on the River Waitiki, New Zealand. The river has the potential for six 90 MW capacity hydropower stations. A form of screen was needed in order to exclude ≥90% of the river's twenty indigenous and four introduced species from flows of >300m^3s^{-1}, making this one of the largest fish screening projects in the world. The screen must exclude both adults and juveniles (25-1000 mm in length) of a range of species including salmonids and eels and be in place all year round. In order to avoid any impingement the maximum contact time has been specified at 60 seconds. The proposed screen gap size is 5 mm with bars orientated vertically.

[15] Meritec Limited, 47 George Street, Newmarket, Po Box 4241, Auckland, New Zealand.

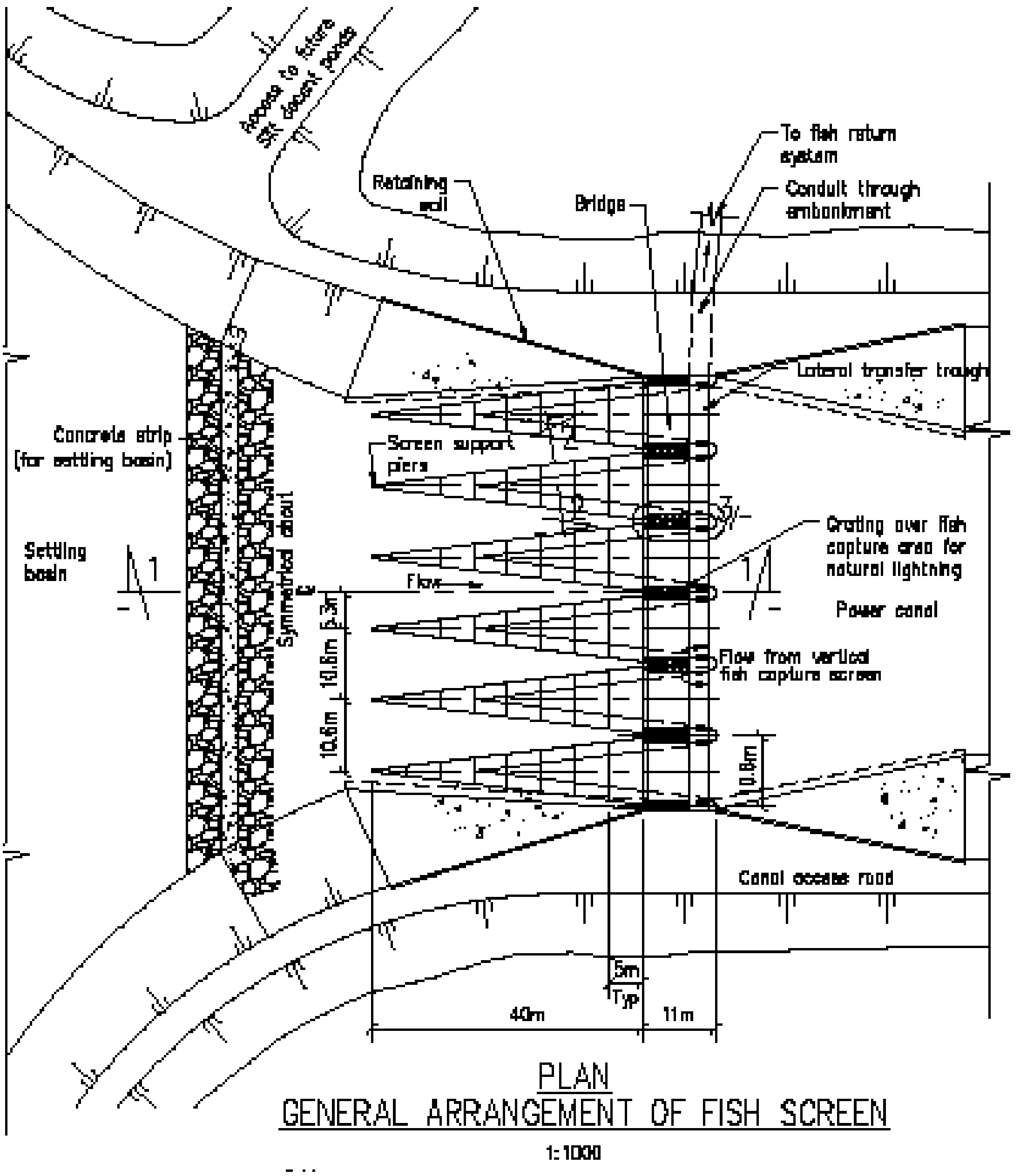


Figure 3.11 River Waitiki arrangement plan for the Labyrinth screen (www.ecan.govt.nz/consents/project-aqua)

The system is based on the 97-98% efficiency seen at the White River labyrinth screen in the USA. This screen is operated at a similar flow and angle as proposed for this installation and successfully excludes chinook salmon fry (*Oncorhynchus tshawytscha*) although using a slightly smaller screen gap of 3.1mm.

The proposed system would consist of wedge-wire screen panels, a collection system and a return system to transport collected fish back to the river. To obtain a low approach velocity the screen would be angled at 8½° to the flow. The labyrinth arrangement confines the screen to a relatively short length of canal making both operation and fish collection easier. A total of 7 labyrinth bays would require 40 m of canal whereas single line vertical screens would require 600 m. A full-height bywash opening and width of 600-900 mm allows fish collection over the full flow depth. Primary screens would consist of bars running perpendicular to the sweeping velocity in order to minimise head loss. An impermeable ramp on the bed angled at 45° ensures accelerating flow into the bywash.

The labyrinth screen concept could be of benefit in the UK at large intakes or where space is at a premium and a compact screening arrangement is required. Low-head hydro would be an obvious application.

3.4 Behavioural Barrier and Guidance Methods

3.4.1 Behavioural Deterrents Background

Deterrent methods are normally used where positive exclusion fish screening is impracticable, owing to the risk of fouling, either by attached biofouling or by waterborne organisms and debris. Fish deterrent systems are commonly known as ‘behavioural barriers’ or ‘behavioural screens’ and are a substitute for, or supplement to, more conventional positive exclusion fish screens. Whereas some positive exclusion screens, when operated and maintained correctly can achieve 100% fish exclusion, behavioural screens cannot.

Fish have a number of well developed senses, and are able to detect and react to light, sound and vibration, temperature, taste and odour, pressure change, touch, hydraulic shear, acceleration, electrical and possibly magnetic fields. The relative sensitivity and capacity to react to any of these stimuli varies with individual species and life stages, each being well adapted to cope with the conditions it is likely to encounter in its particular lifestyle. Environmental variables, such as flow, depth, turbidity, water temperature and others may also affect the success of behavioural methods.

Fish deterrent methods depend on the use of one or more of these stimuli to cause repulsion of fish from the immediate area of the water intake, and in some cases to guide them past the intake into a bywash or to a point downstream. To be effective, the stimulus must be strong enough to repel fish at a range where they are not at risk of being involuntarily drawn in by the strength of the water current. Equally, it must be weak enough to avoid the risk of injuring the fish or of clearing fish from too large an area, which might cause habitat loss and impact upon commercial fishing or block natural patterns of fish migration in rivers.

3.4.2 Louvre Screens

3.4.2.1 Description of Screen

Louvre screens have been used since the 1950s and can be an effective option for the diversion of salmonids and other species. They are in fact a semi-physical barrier which can provide high fish deflection efficiencies (>90%) under optimal conditions (Aitken *et al*, 1966, Solomon, 1992). In general the efficiency of louvre devices varies between 80-100%. In particular high efficiencies have been found for adult and juvenile salmonids as well as American shad (*Alosa sapidissima*), the efficiency is however, lower for alevins and individuals under 5 cm in length. Bottom dwelling fish are not as efficiently deflected, especially where only partial depth louvres are used (Therrien, 2000, Buerkett, 1994, Kynard and Buerkett, 1997).

The louvre screen is based on the reaction of fish to current vortices created by the action of water flow on the louvre slats (Figure 3.12). Approaching fish sense a shearing flow (i.e. different velocities across different points along its body) and as a result avoid the face of the screen. The fish are guided by the angle of the face of the screen into a bywash channel.

For best efficiency slats are positioned at a 90° angle to the incident flow. The individual slats of the screen are spaced at set intervals. The maximum gap used is about 30 cm, suitable for large fish such as adult Atlantic salmon, gaps down to 2.5cm being used for smaller species such as catfish and smelt (Therrien *et al.*, 2000). The angle of the screen to the axis of the flow can vary from 10° to 30° but the optimum is usually found to be between 10° to 15°; efficiency generally decreases as the angle increases. This optimum angle to the flow dictates the length of the screen, which is 3.86 to 5.76 times the channel width (Solomon, 1992). The majority of penetration by fish generally occurs close to the entrance of the bywash and the design is found to benefit from a reduction of slat gaps to around 5 cm close to the bywash entrance. This also reduces the required attraction velocity within the bypass channel. Provided that the slats run to full depth, water depth appears to have little effect on the efficiency of louvre screens and they have been successfully used within a channel depth of up to 4 m (Ducharme 1972).

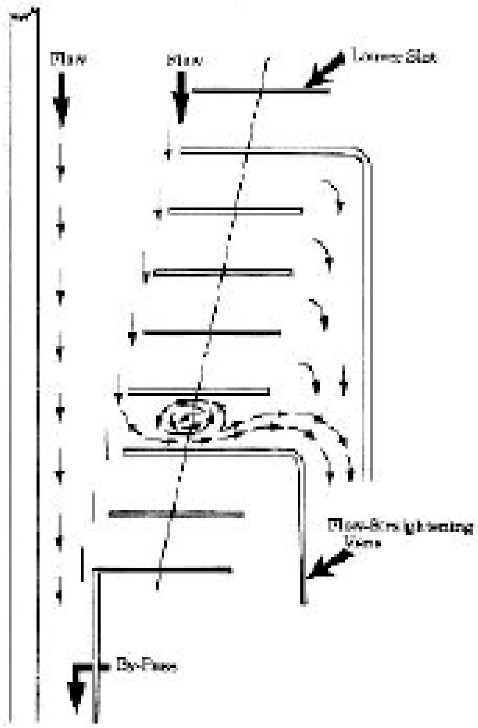


Figure 3.12 Schematic of louvre screen

Although this is a behavioural screen, there is a substantial physical structure involved and debris can become trapped within the channel. This will occur to a lesser extent than conventional mesh screens but regular maintenance will be required and there will be running costs. Cleaning is facilitated if the slats can be lifted away from the screen. Trashing can be reduced by the addition of a coarse trash rack upstream of the louvre screen, which would allow the unhindered passage of fish but prevent movement of debris items. For salmon smolt protection alone it may not be necessary to have the louvre screen in operation all year round and limiting use to the months e.g. of April-May will avoid periods of higher waterborne debris (Solomon, 1992); this, however, would require the negotiation of an exemption under SFFA s.14 unless a local byelaw already allowed this. In some rivers, screening may also be required during the autumn and winter months to protect pre-smolt migrations.

The range of suitable current velocities in the channel has been described in a number of papers and louvre screens have been shown to work from 0.3 – 1.2 ms^{-1} with no loss in efficiency, but they become ineffective as the approach velocity falls below this as shear flows are not generated. The water velocity perpendicular to the face of the louvre array (the 'escape velocity') must be less than the fishes' swimming ability. The velocity in the bywash entrance, however, must be greater than that of the channel in order to provide sufficient attraction. Of velocities tested within the range of 110-300% of the screen approach velocity, a figure of 140% is considered to be the ideal (Solomon, 1992). As the louvre screen itself restricts the water velocity in the main channel a bywash velocity of 1.4 times the main channel is generally easily achieved. At sites where headloss may be a problem the louvres can be fitted with deflectors or current rectifiers along the louvres line at regular intervals to improve hydraulic efficiency (Therrien, 2000).

3.4.2.2 Floating or partial-depth louvres

Floating louvres can be used when screening of just the surface layers is required. This may be the case when screening for fish that only travel in the upper layers of the water column such as salmon smolts, which migrate predominantly in the top 2 m of water. Shad, on the other hand, tend to migrate in the bottom layers. It is believed that with floating devices the current velocity should not exceed 1 ms^{-1} and that the optimum current for which designs are based on is 0.6 ms^{-1} (Therrien, 2000). Ruggles (1990) described a floating louvre screen array fitted at a hydroelectric scheme at Holyoke on the Connecticut River. Based on observations that salmon smolts tend to migrate only in the top metre or two of water, the screen extended over the upper 2.4 m of the 5.5 m water column. The louvre slats were constructed of polypropylene, suspended at 76.2 mm (3”) centres along a 176 m array. This created a screen angle of 15^o across a 44m-wide channel, with a total flow of about 150 m^3s^{-1}. Tests with radio-tagged smolts indicated that 90% were successfully guided into the 4 m wide bypass channel.

Experience of one of the authors (AWHT) at a hydro station in southern Sweden suggests that the success of partial-depth louvres may be dependent on the flow characteristics of the channel. A smolt trap located at the Upper Hemsjo hydro station on the Morrumso River was intended to take advantage of the deflection of smolts swimming near the surface into the trap by an angled ice deflector. This was formed by a metal curtain suspended in the headrace from a floating boom, which was angled across the channel to deflect ice into a bywash chute. The curtain hung a metre or so below the water surface. When very few smolts were found to enter the trap, even though they were observed in the headrace, behavioural observations were made of smolts fitted with float tags (small polystyrene cubes attached to the dorsal fin by a monofilament line) released upstream. It was found that smolts followed the line of the ice boom along most of its length but at a point where it lay nearer to the turbine intake, the flow began to descend towards the submerged intake openings. The smolts were clearly able to detect this descending flow and immediately sounded down below the curtain and passed through the turbine. This observation suggests that partial-depth barrier of any kind are only likely to be effective when flow is uniformly horizontal in direction.

3.4.2.3 Installations in the UK

Very few louvre screens have been installed in the UK, despite extensive testing carried out by the hydroelectric industry in Scotland during the 1960s. There are however, at least two screens known to be installed in Scotland. One is situated at a small, privately owned 100 kVA hydroelectric scheme on the River Almond, a tributary of the River Tay, Perthshire. The screen was installed in the headrace canal to prevent the entrainment of salmon smolts in to the turbine. Such an arrangement benefits from the uniformity of the approach flow, which results in an even hydraulic pattern along the screen face. No testing has been carried out, so efficiency of the screen is unknown.

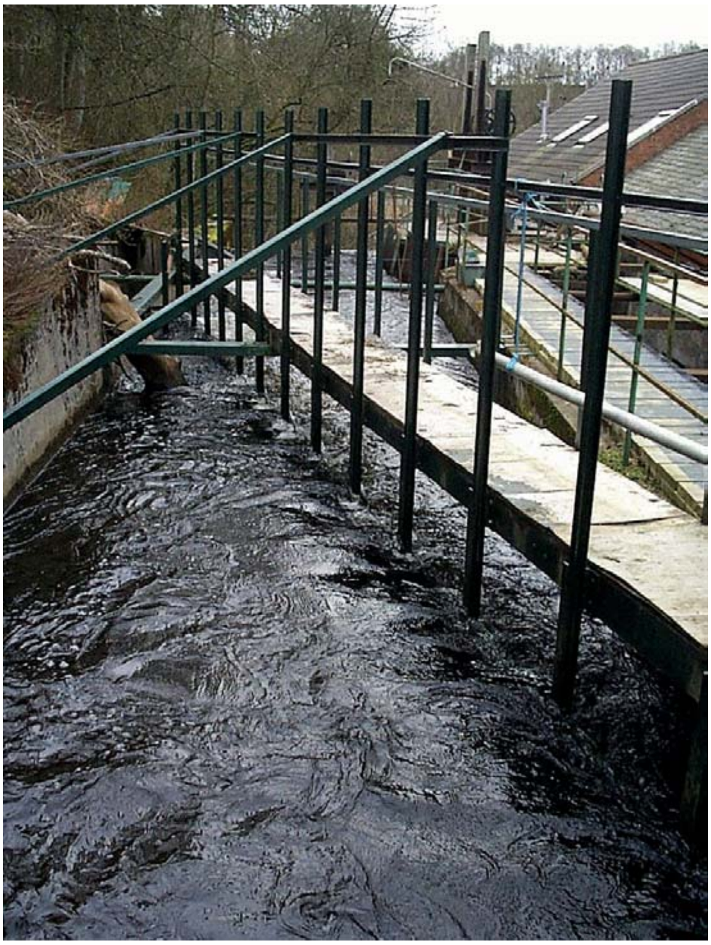


Plate 3.11 The louvre screen at Almond Bank Power Station. Only the superstructure is visible.

A second louvre screen was installed during the 1990s at the 500KVA Blantyre hydroelectric plant on the R. Clyde. This was a partial depth screen, which was not found to be effective, and which was subsequently removed. Unlike the Almond Bank example, it was located directly in the river, as the scheme has no headrace canal. Under these conditions, it is difficult to achieve a uniform velocity profile across the screen.

A form of louvre screen was employed in the 1980s by Thames Water Authority as a smolt trap at Walton water treatment works (Solomon, 1992). The aim of the trap was to provide a means of assessing the efficiency of any other screening device installed at the mouth of the channel. With a cod-end trap in place, bypass acceleration was between 130 and 136%, but when a finer-meshed cod end was installed, flow into the bywash decelerated. With this sub-optimal bywash velocity, it was estimated to be 67% efficient as a smolt trap at maximum discharge, falling to 40-45% at half-flow. Solomon concludes that very much higher efficiencies should have been attainable with increased bywash acceleration.

3.4.2.4 Design and Operational Best Practice

For best performance, louvre screens should be designed with the following characteristics:

- The screen array should be aligned at an angle of 10-15^{o} to the channel axis; the slats should be orientated at 90^{o} to the flow.
- The required slat spacing depends on the size of fish to be diverted, ranging from 30 cm for adult salmon or similarly sized fish, down to 5 cm for juveniles and smaller species. Where smaller slat spacings are needed, they can be arranged so that spacings gradually decrease to the required space along the length of the array, towards the bywash, taking advantage of the reluctance of fish to cross the shear. Flow straighteners should be used to achieve optimal performance.
- Approach velocities should be between 0.3-1.0 ms^{-1} *at all times*.
- Provision should be made for cleaning the louvres, e.g. by having upstream trash racks to catch most of the debris or by having removable slats. Safe access should be provided for this purpose, e.g. via an overhead walkway with safety handrails.
- The screen should run to the full river depth, unless it can be demonstrated that adequate efficiency can be maintained with a partial depth screen (e.g. for surface-swimming fish such as smolts). However, it should not be assumed that all smolts swim at the surface, as smolts tend to sound to the bottom on sensing danger.
- Louvre screens operate best when sited within a headrace canal, or other situations where uniform approach velocities can be achieved. Hydraulic modelling may be beneficial to assess the uniformity of approach flow.
- The bywash entrance design velocity should be around 140% of the screen approach velocity.
- For a more compact arrangement, louvre screens can be arranged in a V-shape (in plan), with the bywash located in the centre (see also Labyrinth screen arrangement (section 3.3).

Low or high velocities will impair screening efficiency, as will any accumulation of debris on the slats. It is important to recognise that they will not prevent fish entry when the water is very slow or static. This means that, for example, on a hydroelectric plant, fish may get past the screen when the turbines are shut down and subsequently be at risk of injury within the turbine(s).

3.4.2.5 Applications

Louvre screens are best suited to canalized waterways where a uniform approach flow can be achieved. They are advantageous over physical screens where large flows must be screened with minimal head loss (e.g. low-head hydroelectric plant). They may be unsuitable for locations subject to inundations of weed, e.g. on chalk streams where weed cutting is carried out.

3.4.2.6 Fish Species/Lifestages

Suitable for salmonid smolts and adults, adult shad, and probably most non-benthic species. Louvre screens are not suitable as fry screens, as the slat spacing would be impracticably small. For adult shad exclusion, the screen would need to be full depth.

3.4.2.7 Ease of retrofitting

Louvre screens are suitable for retrofitting into engineered channels, e.g. hydroelectric headraces or water supply aqueducts. A trash rack placed upstream will assist with debris clearance.

3.4.3 Bubble Curtains

Bubble screens are one of the most basic behavioural barrier types. This form of screen works on the principal of a curtain of bubbles being generated via a perforated tube laid along the riverbed through which compressed air is pumped. The wall of bubbles is usually laid at an angle to the flow, or in a loop around the intake entrance and is used to deflect approaching fish and guide them either into a bywash (*cf.* louvre screen arrangement) or to a point downstream of the intake. The exact nature of the deterrent effect is uncertain and may be due to a combination of visual, auditory or shear-current stimulus (Solomon, 1992).

Plate 3.12 Bubble curtain laid across a small stream

Turnpenny (1998) suggested that from personal experience that bubble curtains work at highest efficiency in flowing channels and when placed at a slight angle to the bank (~12°). This relies upon glancing contact with the fish in order to deflect them across the channel.

Aspects of the design that can effect the efficiency and performance of the screen include the size and spacing of bubbles, volumes of air discharged, air pressure, water velocity, screen layout and illumination (Solomon, 1992).

Several investigations have been carried out over the efficiency of these screens over the last 60 years. One of the first laboratory experiments was carried out in 1942 and found mixed success with different species (Bramsnaes *et al*., 1942). Whilst carp (*Cyprinus carpio*) and pike (*Esox lucius*) were deflected by the screen rainbow trout (*Onchorhynchus mykiss*) were not deterred and passed freely. Many investigations have shown inconclusive results although some have shown a high success rate. A deflection rate of up to 98% was recorded during British Columbian and Ontario Hydro experiments (Brett and MacKinnon, 1953, Patrick et al., 1985) although falling to 80-51% during darkness. This would suggest that a stimulus of reflected light is partially responsible for the screens deflection effects. These results of high deterrent abilities must be looked upon cautiously due to often mixed and inconclusive results from other investigations. Laboratory tests also generally do not allow for extended periods of continuous screen use in which time fish become habituated to the screen stimulus, although this effect is only likely to apply to resident populations rather than actively migrating fish. Field investigations, although few, have resulted in even more mixed opinions over efficiency and have in general resulted in lower efficiencies than laboratory investigations.

3.4.3.1 Installations in the UK

Solomon (1992) reports on bubble curtain trials carried out at the experimental installation at Walton water treatment works on the R. Thames. This comprised six 4m lengths of 50mm diameter galvanized pipe drilled with 2mm diameter holes at 25mm centres along the length. Air was supplied by a blower rated at $348m^3h^{-1}$ discharge @1 bar pressure. Water depth was around 2m. Fish entrainment was compared by monitoring catches in the louvre-screen trap (see above) with and without the bubble curtain operating. The bubble curtain was also operated in conjunction with an array of nine submerged strobe lights flashing at 440 flashes per minute. On four of six occasions when the bubble screen was operated alone, fish entrainment was less than predicted; when both the bubble screen and strobes were operated, entrainment was reduced by an estimated 62.5%. Overall, it was estimated to have reduced entrainment of smolts from 14.4% of the total run to 5.4%.

Experiments carried out on a 70 m-long bubble curtain placed across the entrance to the cooling water intake at Heysham Power Station (Lancashire) resulted in a reduction of fish entrapment by 37% which was significant at the $P<0.001$ level (Turnpenny, 1993). This was an improvement on the previous situation, probably saving some tonnes of juvenile fish each year. Catch rates on the cooling water drum screens were compared for alternating six-hour periods with bubble curtain on or off, and in daylight versus darkness. The bulk of the fish catch comprised sprat (*Sprattus sprattus*) and herring (*Clupea harengus*) but 42 fish species were recorded during the trials, which took place over a 24-day period during the month of February. The bubble curtain also reduced entrainment of brown shrimps (*Crangon crangon*) by 56%. An unexpected outcome of the trials was that the curtain was more effective at night. This was attributed to the nocturnal behaviours of clupeid fishes and shrimps, which tend to disperse vertically into the water column at night. Repulsion was considered by the author to be partly related to physical effects of the rising bubbles and induced currents.

In static and slow-moving conditions bubble curtains are less effective (Turnpenny, 1998). Use of bubble curtains was attempted by the Environment Agency at the intake of the Blackdyke Pumping Station in Lincolnshire. The water supply channel is virtually static and although results were initially positive the success rate reduced over the following weeks, presumably as fish habituated to the stimuli (Turnpenny, 1998). On the other hand fast, turbulent or deep waters can lead to break-up of the bubble sheet with loss of efficiency. This is important as performance may deteriorate at the most critical time for fish that migrate on floodwaters. The maximum reliable depth for a bubble curtain is about 3 m; above this, the bubbles tend to form cords, which split apart, leaving gaps. This could potentially be overcome by placing bubble pipes at height intervals of ≤3 m in the water column but this is not straightforward and puts the pipes at risk of damage by flood debris.

To achieve the most effective performance from a bubble curtain a strong flow of air must be used. The most economical way of generating the air supply depends on the flow rate, depth and application of the specific project. When used in less than 2m of water a simple, low pressure rotary blower gives economical and reliable mechanical performance, whereas for greater depths a multi-stage blower or air compressor may be needed to overcome the greater hydrostatic pressure (Turnpenny, 1998).

Before installing a bubble curtain certain behaviours of bubbles must be taken into account. Bubbles larger than 2mm in size will rise through the water at a rate of approximately $0.25ms^{-1}$; smaller bubble sizes are not recommended in moving water, as they rise too slowly. Before installation, the surfacing line of the bubbles should be calculated from the velocity of the water in order to determine the correct positioning of the barrier on the bed. Where a bywash is used, the width of the mouth must be able to accommodate any variation in surfacing position, otherwise fish may not find the entrance. It may be necessary to have more than one bubble pipe in order to accommodate any changes in flow conditions. The bubble curtain itself will also create a certain degree of turbulence and may therefore require some fine-tuning of airflow to achieve a uniform curtain of bubbles (Turnpenny, 1998).

A bubble curtain was recently installed at the entrance to the cooling water intake at Fawley Power Station (Hampshire). An acoustic fish deterrent (AFD) system was also installed. The application is unusual, since the curtain is not intended to deflect fish directly (although it may have some benefit in this respect) but to help prevent a build up of silt in the channel, which might increase the intake velocity and impair the propagation of sound from the AFD.

3.4.3.2 Bubble screens in combination with other behavioural stimuli

Combinations of bubble curtains with other types of behavioural screens generally achieve greater efficiencies than when used alone. The best combinations involve adding acoustic and or artificial light stimuli.

A combination of bubble curtains and strobe lights used at Walton-on-Thames has been described above. Another was tested by Sager *et al*. (1987). Very low efficiencies were found for bubble curtains alone with very little effect being seen in all species. On combining bubble screens with strobe lights at a flash rate of 300 min^{-1}, up to 100% efficiency was seen in spot (*Leiostomus xanthurus*), 68% for menhaden (*Brevoortia tyrannus*) and 36% for white perch (*Morone americana*).

The second option involves the combination of a sound generator with a bubble sheet creating a 'wall of sound' that can be used to guide fish into a bywash. The system is known as a 'Bioacoustic Fish Fence' (BAFF™) and is designed by Fish Guidance Systems (FGS). Further details of light and acoustic barriers are given below.

3.4.3.3 Design Best Practice

Key points in good bubble curtain design and operation are:

- ensure that a uniform bubble sheet, free of large gaps, is maintained under all flow conditions where fish exclusion is required; hole spacings in bubble curtain design are considered roughly analogous to bar separations in bar screens, so do not expect 15 cm hole spacings to exclude smolts! Small holes of 0.5-2 mm bore spaced at 1-3 cm are usually effective.
- allow for plenty of air flow: at least 1 ls^{-1} per metre of barrier length and up to 4 ls^{-1};
- estimate the velocity profile along the proposed screen line for the range of expected conditions and calculate the surfacing lines under various scenarios, making sure that the curtain leads into the bywash entrance (if applicable);
- select a position where the depth along the line is as uniform as possible to avoid loss of air flow in deeper areas; avoid areas where the bed is unstable;
- for best results, angle the curtain near-parallel to the flow and preferably at an angle of more than 15^{o} to the channel flow;
- check the bubble pipe regularly and keep it clear of bed materials and biofouling; blockage is more likely to occur if the curtain is not operated continuously; check for uniformity of the surface air plume.
- warning systems (e.g. via telemetry links or visual inspection) should be provided to inform plant operators of air supply failure.
- The equipment requires regular maintenance and service intervals should be displayed and logged in the plant control room.
- Back-up power or interlocks with pump controls may need to be provided to ensure that pumping does not occur when the system has lost power.

3.4.3.4 Applications

Bubble curtains may be used as a low-cost behavioural barrier in flowing water situations where high performance is not demanded. Fast-flowing or deep water may lead to an unacceptable breakup of the curtain's integrity, reducing effectiveness.

3.4.3.5 Fish Species and Lifestages

Many fish species, including UK salmonids, clupeids and cyprinids can be deflected by a bubble barrier but habituation is rapid. Consequently they are best suited to deflection of migrating fish in rivers, or of fish moving with the tide in tidal systems. In these situations contact time is likely to be short. Bubble curtains alone are probably ineffective for eels and lampreys, but the addition of artificial lights or strobe lights enhances their efficiency for eel deflection (see below). There is a risk, however, that illumination may attract some species, for example 3-spined stickleback (*Gasterosteus aculeatus*) (Hadderingh, 1982).

3.4.3.6 Ease of retrofitting

Bubble curtains can easily be retrofitted to almost any existing application.

3.4.4 Electric Barriers

Electric intake screens were first developed in the 1950s by MAFF Fisheries Laboratory. After their development several were installed across the UK but most were later removed over fears of their safety. The effectiveness of such screens is uncertain and has historically been thrown into doubt. As well as uncertainty over the efficiency there are also concerns about safety and the risks that they may pose to both animals and humans, although that is not to say that all electric screens are inherently unsafe.

A critical issue with electric screens is that the potential difference experienced by a fish is dependent upon the source voltage and the size of the fish. Larger fish are exposed to a proportionately greater voltage than smaller fish. The electric field must be strong enough to repel small fish but at the same time may be too strong for larger fish, stunning them and causing them to be drawn into the intake (Turnpenny, 1998).

The MAFF electric screen was an array of vertical electrodes set approximately 15-30cm apart and of alternating polarity. They were arranged across the intake entrance throughout the depth of the water column. Upon energising, a local electric field is created designed to repel fish.

More recently a USA company[16] has developed a newer version of the electric fish screen called a Graduated Field Fish Barrier (GFFB™), which claims to be both safer and more effective than traditional designs. The GFFB™ uses direct current (DC) which is less stressful to fish than an alternating current (AC). Short pulses energise a parallel array of electrodes. To produce the most effective electric field for fish deterrence it is desirable for the electric lines to run from head to tail along the fish. As fish instinctively swim with their head into the flow the most efficient design is to therefore have electric field lines running parallel to the water flow. When the fish is crosswise to the field it will receive no shock.

The most important feature of the GFFB™ is the graduated field itself. An increasing voltage field is produced along the array. This results in larger fish being affected by the electric field at an early stage of the array and gradually smaller and smaller fish are affected as they penetrate further into the array. Large fish turn and are carried out or swim away from the intake before they are stunned and smaller fish are deterred at a later stage.

The GFFB™ is supplied in versions for upstream or downstream guidance (see also section 3.4). For downstream guidance the system differs in that it has an abrupt leading field edge designed to invoke a startle reaction in the fish causing them to dart away from the array. The fish are guided into a bywash system by angling the array in relation to flow (Figure 3.13).

[16] Smith root Inc., 14014 NE Salmon Creek Avenue, Vancouver, WA 98686, USA.

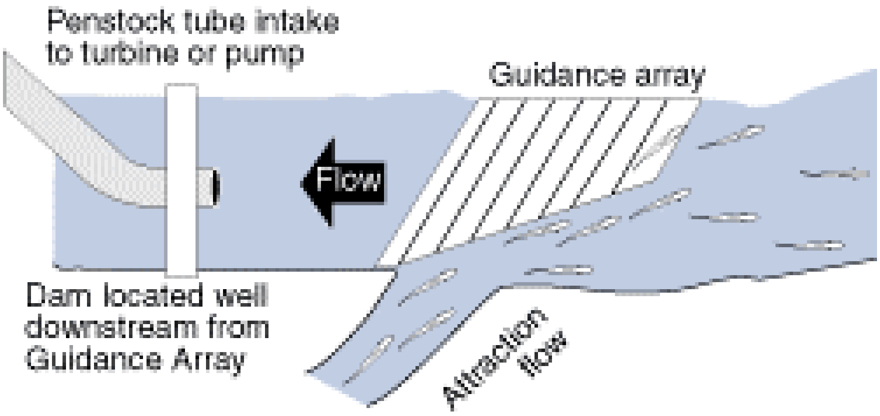


Figure 3.13 Diagram of the GFFB™ in use as a downstream guidance system (www.smith-root.com). Note that fish often turn to face the flow when confronted with a barrier.

Safety to humans is obviously a concern as an electric field is still present in this design. The manufacturers claim that the short electrical pulses used by this system are much less hazardous to humans. Safety can, however, be further improved by limiting public access, a course of action which is recommended.

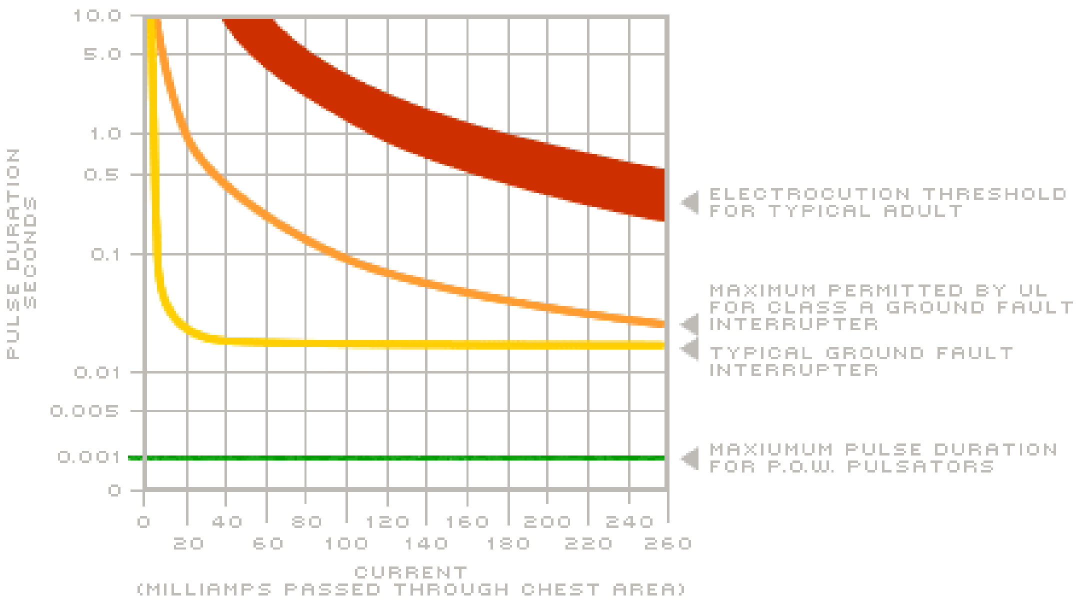


Figure 3.14 Electrical pulse duration and current data for the GFFB™ pulse generator in relation to human risk criteria (www.smith-root.com).

Hilgert (1992) carried out investigations on both the effectiveness of the system and long and short-term physiological impacts on adult salmonids and their gametes. A test on the potential injury to pre-spawning adults resulted in no injuries or mortality to adult coho salmon (*Oncorhynchus kisutch*) after exposure to an electric field of 0.2 to 0.9 Vcm^{-1} for 10 seconds. It was also concluded that there was no effect on gamete viability or early development after exposure of up to 0.9 $V.cm^{-1}$ for 10 seconds. The effectiveness of the system as a barrier was monitored at the Quilcene National Hatchery and although the system was generally successful fish did pass during high water conditions. The problem has since been overcome by the addition of an automatic pulse-width control, which regulates the electrical output in relation to flow conditions.

Barwick and Miller (1994) attempted to simulate the downstream migration conditions in a hydroelectric headrace canal. A variety of North American fish (salmonids and clupeids) were introduced to the canal whilst the GFFB™ was operated at 10 pulses per second. The percentage of fish not passing the barrier while water was flowing was 83% with an electric field strength of 1.5 $V.cm^{-1}$; this rose to 97% when the water was static. These results are encouraging but it should be noted that the size of the canal was comparatively small, being 2 m wide x 1 m deep.

Results of the Graduated Field Fish Barrier seem to be more promising than the early MAFF-designed electric barrier although further testing is still necessary.

A German company, Geiger International GmbH (website: http://www.geiger-international.de) also markets an electric fish screen, known as the Fipro-Fimat Fish Repelling Device. It uses a randomized electrical pulse generator, with the stated purpose of reducing habituation to the electrical signal. The system is intended to be detected by fish some 5-10m upstream of the electrode array. No information is given on the website about the effectiveness of the system as a fish barrier, nor concerning health and safety aspects, nor of any existing installations of the system.

3.4.4.1 Installations in the UK

Other than perhaps the odd MAFF-type screen remaining in place, no electric intake barrier installations are known of in the UK. Installations at outfalls are discussed below (section 3.6).

3.4.4.2 Design Best Practice

Electric screens in general are not recommended for intake screening. The GFFB™ may be more suitable, owing to its use of a graduated field, which should, in theory, lessen the risk of larger fish becoming stunned while smaller fish remain insensitive to the field. There has been no scientific testing of the GFFB™ for intake screening and therefore its performance is unknown and it is premature to discuss 'best practice'. Where testing is contemplated, the manufacturer's recommendations should be followed. Warning systems (e.g. via telemetry links) should be provided to inform plant operators of failure. In general for electric screens (for intake and outfall applications), the following points should be noted:

- The equipment should be regularly maintained and service records should be displayed in the plant control room.
- Visible indicators of the operational status of the electric screen should be displayed at or close to the intake to inform operational and enforcement personnel.
- Back-up power or interlocks with pump controls may need to be provided to ensure that pumping does not occur when the system has lost power.

3.4.4.3 Applications

Electric barriers are affected by water conductivity and are unsuitable for marine or brackish water environments.

3.4.4.4 Fish Species and Lifestages

Electric barriers are best suited to the deflection of large fish, as relatively low, safe voltages can be used; conversely, it is unlikely that small fry could be protected without using excessively high voltage. The GFFB™ would appear to lend itself to diversion of eels and lampreys, being elongate species that swim close to the riverbed. This merits further testing in the UK.

3.4.4.5 Ease of Retrofitting

The traditional MAFF-type electrode array was a very simple retrofit to almost any kind of intake. The GFFB™ is less straightforward to retrofit, as the electrodes must be attached to a flat bed/walls of insulating material. Nevertheless, this is not usually an insuperable civil engineering task.

3.4.5 Acoustic Guidance

The hearing range of most fish falls within the audible range to humans, maximum sensitivity lying in the sub-3 kHz band down to infrasound frequencies (Hawkins, 1981; Sand and Karlsen, 1986). Acoustic fish deterrent (AFD) systems mostly exploit hearing sensitivity in the 20 to 500 Hz range, although infrasound (<20 Hz) and ultrasound (usually >100 kHz) systems have been used with some success (Knudsen *et al*., 1992, 1994, 1997; Carslon, 1995; Turnpenny *et al.*, 1998; Sand *et al.*, 2001). The usefulness of ultrasound in this context appears to be limited to guidance of clupeid species, which have auditory sensitivity at these frequencies, possibly an evolutionary adaptation to evade cetacean predators (Mann *et al.*, 1997).

Early work in this field was by American researchers Loeffelman *et al.* (1991a,b) and Klinect *et al.*(1992), who discovered that underwater machinery noise emitted by bulb turbines at Racine hydroelectric plant (Columbia River, USA) caused fish to avoid areas close to the turbine intakes. Bulb turbines differ from most designs in that the generating machinery is submerged. These researchers investigated acoustic repulsion further and developed and patented a method of signal development, based on recording and analysing fish communication sounds. The process involves the spectral analysis of fish sounds, followed by the synthesis of a signal containing key elements of the spectrum. The synthesised sound signals were then amplified electronically and generated underwater using military sound projectors. Field trials showed that significant fish avoidance could be achieved using this technology, sparking interest in the method for applications in the UK. The Energy Technology Support Unit (ETSU), Harwell, funded work initially to establish whether and how the technique could be applied to fish protection at tidal power schemes. The resulting collaborative study with the American team (Turnpenny *et al*., 1993) demonstrated that repellent signals could be developed for European fish species, although it was shown that a more empirical method of signal development than that proposed by Loeffelman was more cost-effective. The species studied included Atlantic salmon, trout (*Salmo trutta*) and various estuarine species. Subsequent experiments have found signals that are effective against other fish, including Twaite shad, most cyprinid and percid species and a wide range of marine and estuarine fishes. In recent years there have been considerable advances in the field of acoustic fish guidance and sound-based systems are now widely used and validated. Nevertheless, the apparent failure of acoustic methods in various scientific trials (see e.g. Turnpenny *et al*., 1994; Goetz *et al*., 2001) highlights the fact that this is not an easy or universally suitable technology.

Plate 3.13 Acoustic sound projectors being prepared for installation at Amer Power Station (The Netherlands) (courtesy Fish Guidance Systems Ltd)

3.4.5.1 Sound Signal Characteristics

AFD sound signals need to have maximum effect for minimum energy input. Low frequency (LF) sound (10 Hz – 3kHz) is used for all species other than clupeids; for clupeids either low frequency or ultrasound can be used for good results.

Two main methods of generating a LF acoustic barrier are presently in use in Europe. One, known as the SPA™[17] (Sound Projector Array), uses arrays of underwater transducers or "sound projectors" to produce a diffuse field of sound that will block fish movement. The other, known as the BAFF™ (Bio-Acoustic Fish Fence) employs sound sources coupled to a bubble curtain (see also discussion of bubble barriers above) to produce a discreet "wall of sound" that can be used for more precise guidance of fish, e.g. into a bywash channel. The BAFF™ system is used primarily for diversion of fish into bywash channels rather than for blockage.

Benefits of using LF sound signals are:

- Low-frequency sound (unlike ultrasound or light) penetrates even the most turbid waters.
- Detection of sound and vibration is one of the primary sensory modalities in fish, especially in waters of low transparency; most fish are sensitive to LF sound.

[17] Fish Guidance Systems Ltd, Belmore Hill Court, Owslebury, Winchester, Hants SO21 1 JW. Website: www.fish-guide.com

Lambert *et al.* (1997) identified the following key signal characteristics for a LF SPA system:

1. The sound signal should be within the frequency spectrum 10 Hz – 3 kHz.
2. The nature of the signal should be repellent to fish. Pure tones do not deter fish, except at very low frequencies that are difficult to generate (e.g. 10 Hz) or at very high sound pressure levels, which are expensive to generate. The most cost-effective deterrent signals use either a blend of different frequencies applied as a pulse or crescendo, or a 'chirp' comprising sweep across a frequency band.
3. The sound level received by the fish at the required point of deflection should be sufficiently above ambient noise level (typically at least ten times, or >20dB), although this depends on the species of fish and the type of signal).

More recent investigation helps to clarify the last of these points. Nedwell *et al.* (in press) have proposed that the degree of reaction to sound in fish cannot be predicted from just the received sound level and the background noise level without knowledge of the hearing sensitivity of the fish, as expressed by an audiogram (plot of hearing sensitivity on a decibel or dB scale versus sound frequency). Based on field trials, they propose the following approximate levels in relation to fish behaviour; the levels shown are the peak sound pressure levels calculated when the audiogram values are subtracted from the received noise spectrum and are known as $dB(ht)_{species}$ levels[18]:

Sound Level ($dB(ht)_{species}$)	Fish Behaviour
+30 dB	Threshold of visible reaction in more sensitive individuals
+50dB	Most fish swim away from the sound
+70dB	Strong aversive reaction.

While this advice is an advance, it is not complete, as the shape of the sound signal also influences the degree of fish reaction (Turnpenny *et al*., 1983).

3.4.5.2 SPA System Hardware

A SPA acoustic deflection system comprises the following components, arranged as shown in Figure 3.16:

- an electronic signal generator,
- one or more power amplifiers,
- an array of underwater sound projectors,
- inter-connecting cables.

[18] 'ht' stands for 'hearing threshold'; the subscript 'species' represents the particular species; thus, $dBht_{salmon}$ measures the peak level of a sound emission as heard by a salmon.

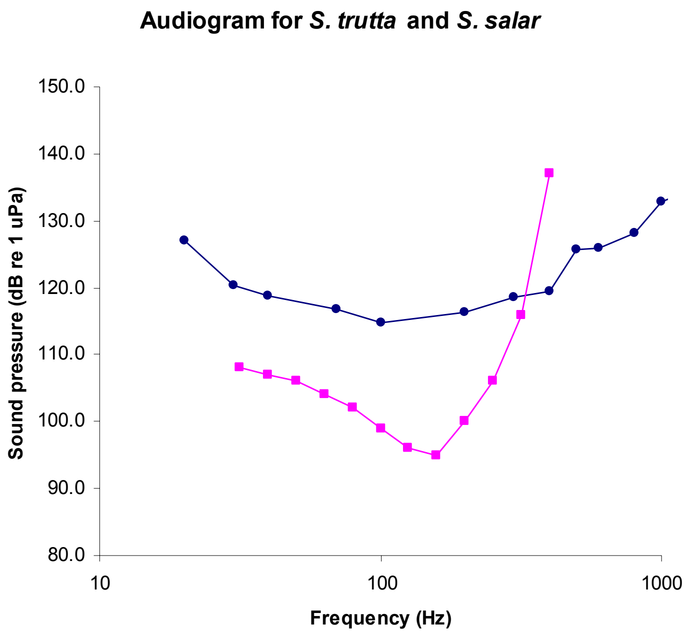


Figure 3.15 Audiograms of Atlantic salmon (*Salmo salar*) (■) and trout (*S. trutta*) (●) (Nedwell *et al*., 2003).

The system is analogous to a public address or domestic hi-fi system. The signal is usually recorded onto an EPROM-chip and the signal generator may contain a number of these, which can be manually selected or played at random or in rotation. One or more high-power audio amplifiers that are matched and filtered to suit the sound projectors amplify the signal. The sound projectors are underwater transducers, analogous to loudspeakers. These are electromagnetic devices, with a piston-type arrangement connected to a rubber diaphragm.

As there is an air cavity behind the diaphragm, sound projectors are susceptible to pressure change, increasing pressure tending to force the diaphragm inwards. This limits the throw of the diaphragm and must be compensated by a balancing pressure from behind. Sound projectors are therefore either pre-pressurised to cope with the expected operating depth, or else have some form of pressure compensation device. The latter type is best suited to use at fixed positions in tidal waters.

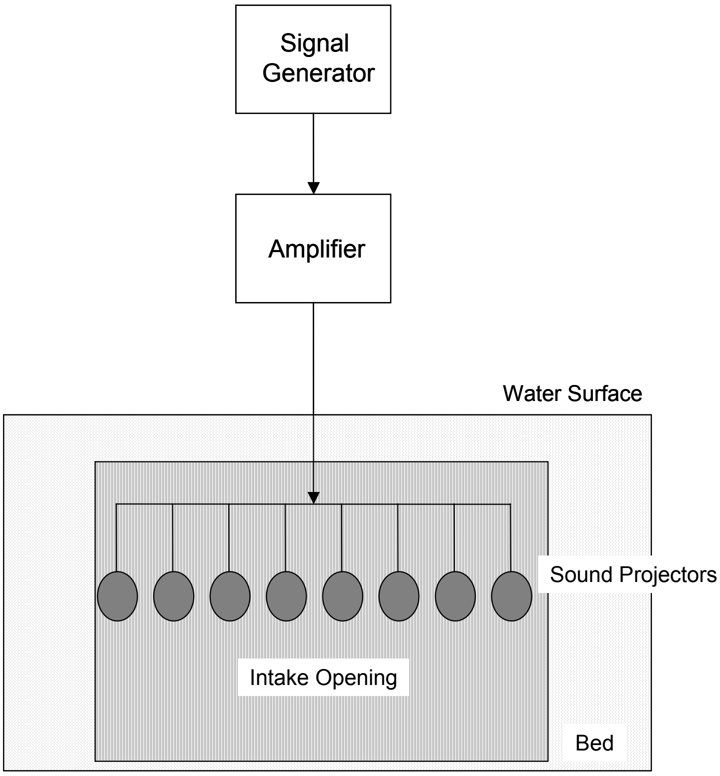


Figure 3.16 Schematic layout of SPA AFD system (intake elevation)

3.4.5.3 Acoustic Field Design

For best results, the sound projectors are located close to the intake opening, so as to yield high signal particle velocities in the paths of incoming fish. The optimum number and positioning of sound projectors can be determined using an acoustic model such as PrISM™[19] to predict the resulting sound pressure and particle-movement field (see example in Figure 3.17). The PrISM model also accommodates information on the geometry and bathymetry of the intake area and adjacent structures, and ensures that surface and bed reflections are taken into account in the final system design.

The ideal sound field should form a steep acoustic gradient approaching the entrance, free from acoustic nulls caused by destructive interference within the sound field. The presence of such nulls could cause fish to be guided into, rather than away from the intake (Lambert *et al.*, 1998). After commissioning, measurements can be taken to confirm the field characteristics and to ensure that there is no risk of deterring fish over too large an area.

[19]Subacoustech Ltd, Chase Mill, Bishops Waltham, Hants, SO32 1AH

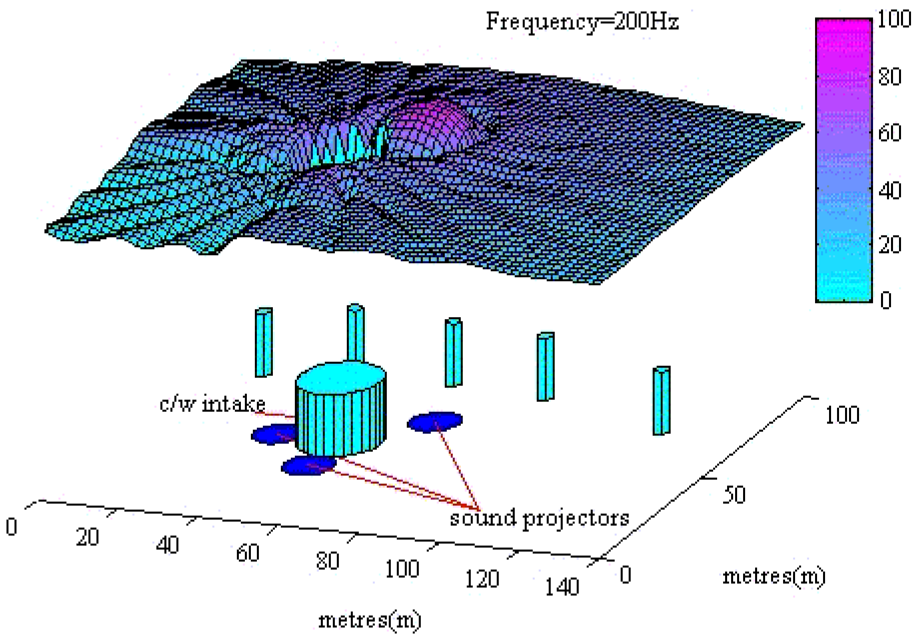


Figure 3.17 PrISM acoustic model of the acoustic deterrent field at Doel nuclear power station, Belgium (Subacoustech Ltd).

The lower part of the diagram shows the positions of the sound projector clusters around the intake caisson. The five pale blue columns are dolphins that protect the offshore side of the intake from damage by shipping. The upper part of the plot shows the acoustic field at 200 Hz, represented in units of dB re 1μPa. In practice, the model is run for a variety of tidal heights and over the full range of signal frequencies to be used.

3.4.5.4 Effectiveness of SPA Acoustic Deterrents

Turnpenny *et al*. (1998) showed that well designed LF SPA acoustic deterrent systems offer amongst the highest deflection efficiencies available from any type of behavioural barrier. Values of up to around 80% can be attained for many teleost species possessing a developed swimbladder, with recorded values of between 90% and 100% for the most sensitive species such as herring. Table 3.2 shows test results from various installations that have been studied.

The longest-running trials of a SPA system have been conducted at Doel Nuclear Power Station on the Zeeschelde Estuary (Belgium). Since the installation of an FGS SPA system in 1997, regular monitoring has been carried out by Leuven University (Maes *et al*, 2004). As in most trials of this kind, comparisons have been made of the fish impingement rates on alternate days with the sound system turned on or off. Although the species captured have been primarily of estuarine or marine origin, quantities of freshwater fishes have been caught.

As with other behavioural systems, habituation to the stimuli must be considered. Habituation, again, is not a problem with migratory or highly mobile fish, which are rarely in contact with the sound for a long period. Nevertheless, it is an aspect relevant to resident fish populations, where fish may be in contact with the sound for extended periods. Acoustic deterrent signals are developed specifically to minimise the risk of habituation over a period of a few days at least (Turnpenny *et al.*, 1993), but for more extended exposure the deterrent signal may need to be altered at intervals (e.g. once per day). Signal generators with multi-signal capability may be used for this purpose.

Table 3.2 Results from Acoustic Barrier Trials

Location	Fish Species	Diversion Efficiency, Significance Level	Reference
R. Foss flood relief pumping station, York (32 $m^3.s^{-1}$) [Freshwater- river]	Chub (*Leuciscus cephalus*) Roach (*Rutilus rutilus*) Bleak (*Alburnus alburnus*) Bream (*Abramis brama*) Perch (*Perca fluviatilis*) All species	87% P<0.02 68% P<0.001 72% P<0.05 74% P<0.05 56% P<0.05 80% P<0.001	Wood *et al*., 1994
Hartlepool nuclear power station (34 $m^3.s^{-1}$) [marine]	Herring (*Clupea harengus*) Sprat (*Sprattus sprattus*) Whiting (*Merlangius merlangus*) Other swimbladder fish Non-swimbladder fish	79% P<0.01 60% P<0.05 54% P<0.05 55% P<0.05 16% P>0.05	Turnpenny *et al*., 1995; Turnpenny & Nedwell, in press.
Blantyre Hydro-electric plant (20 $m^3.s^{-1}$) [Freshwater - river]	Salmon (*Salmo salar*) Mixed cyprinid species	74% P<0.02 92% P<0.02	Anon., 1996
Farmoor Water Supply Intake [Freshwater - river]	Coarse fish, mainly perch (*Perca fluviatilis*)	 87% P<0.02	Turnpenny *et al*., 1998
Doel 3 & 4 Nuclear Power Station [Estuarine]	Herring Sprat Smelt (*Osmerus eperlanus*) Bass (*Dicentrarchus labrax*) Flounder (*Platichthys flesus*) Gobies (*Pomatoschistus spp.*) Crustaceans	95% P<0.001 88% P<0.001 64% P=0.004 76% P<0.001 38% P<0.05 46% P=0.028 50% P>0.05	Maes *et al*., 2004

3.4.5.5 SPA Maintenance and Monitoring Requirements

Sound projectors are electro-mechanical devices and regular maintenance of them is required to maintain optimum performance. This involves removing the underwater units to replace perished seals and to check moving components. Also, it is desirable to raise and clean the units occasionally to remove any build-up of silt or fouling. It is essential that some mechanism be provided to bring sound projectors to the surface for maintenance, without the need to use divers.

As it is difficult to check the performance of submerged equipment, diagnostic units can be attached to the shore-based electronics to monitor performance of the sound projectors and associated electronics. These can be linked by telemetry systems to control centres in the case of remote sites. Performance of the systems can then be electronically logged and made available to regulatory enforcement staff.

3.4.5.6 Potential Public Noise Nuisance

Although emitting frequencies that are within the human audible range, the location of the sound projectors below water generally prevents any audible acoustic propagation into the air above. SPA systems are occasionally just audible from at the intake position under exceptionally quiet conditions, particularly if they have been mounted on any metallic structures that project out from the water. A SPA system operated by the Environment Agency at the River Foss flood relief pumping station in York is located <50m from a city-centre hotel and has operated during pumping since 1995 without complaint.

3.4.5.7 Installations in the UK

SPA AFD systems are widely used in the UK, with examples at locations shown in Table 3.2. Also shown is other information about the specifications of the systems, the operating environment and the types of fish to be protected.

Table 3.3 Examples of Sound Projector Array AFD Systems Installed in the UK (source: Fish Guidance Systems Ltd)

Application	Location & Max. Intake Flow	Main System Components	Main Fish to be Protected
Estuarine Power Stations	Great Yarmouth, Norfolk $9.3m^3s^{-1}$	Signal generator, 8 x large sound projectors, 8 x 450W amplifiers	Estuarine & marine fish, mixed
	Fawley, Hampshire $31m^3s^{-1}$	Signal generator, 8 x large sound projectors, 8 x 450W amplifiers	Estuarine & marine fish, mixed, salmon & sea trout
	Shoreham, West Sussex $5.6m^3s^{-1}$	Signal generator, 6 x large sound projectors, 6 x 450W amplifiers	Estuarine & marine fish, mixed & sea trout
Potable Water Intakes	Surbiton, R. Thames $2.7m^3s^{-1}$	Signal generator, 8 x small sound projectors, 1 x 450W amplifiers	Salmonids and mixed coarse fish
	Laleham, R. Thames $12m^3s^{-1}$	Signal generator, 8 x small sound projectors, 1 x 450W amplifiers	Salmonids and mixed coarse fish
	Hythe End, R. Thames $3.2m^3s^{-1}$	Signal generator, 8 x small sound projectors, 1 x 450W amplifiers	Salmonids and mixed coarse fish
	Datchet, R. Thames $24m^3s^{-1}$	Signal generator, 8 x small sound projectors, 1 x 450W amplifiers	Salmonids and mixed coarse fish
	Walton, R. Thames $14m^3s^{-1}$	Signal generator, 8 x small sound projectors, 1 x 450W amplifiers	Salmonids and mixed coarse fish
	Hampton, R. Thames $5.8m^3s^{-1}$	Signal generator, 8 x small sound projectors, 1 x 450W amplifiers	Salmonids and mixed coarse fish
	Farmoor, R. Thames $2.7m^3s^{-1}$	Signal generator, 8 x small sound projectors, 1 x 450W amplifiers	Mixed coarse fish
	Canaston, W. Cleddau 0.70 m^3s^{-1}	Signal generator, 8 x small sound projectors, 1 x 450W amplifiers	Salmonids
	Kilgram Bridge, R. Nidd 0.54 m^3s^{-1}	Signal generator, 4 x large sound projectors, 4 x 450W amplifiers	Salmonids and mixed coarse fish
	Barcombe, R. Sussex Ouse, 0.845 m^3s^{-1}	Signal generator, 4 x small sound projectors, 1 x 450W amplifiers	Salmonids and mixed coarse fish

3.4.5.8 *Evanescent Sound: The BioAcoustic Fish Fence (BAFF™)*

An evanescent (non-propagating) sound field is one that decays rapidly with distance from its source. The Bio-Acoustic Fish Fence (BAFF™) is a proprietary product of Fish Guidance Systems Ltd (FGS) of Southampton, England that uses a combination of a sound source and a bubble curtain to create a field that is largely contained within the bubble sheet (Nedwell and Turnpenny, 1997). Physically, it comprises an electromagnetic or pneumatic sound transducer coupled to a bubble-sheet generator, causing sound waves to propagate within the rising curtain of bubbles. The sound is contained within the bubble curtain as a result of refraction, since the velocity of sound in a bubble-water mixture differs from that in either water or air alone. The sound level inside the bubble curtain may be as high as 170 dB re 1μPa, typically decaying to 5% of this value within 0.5-1 m from the bubble sheet (Figure 3.18). It can be deployed in much the same way as a standard bubble curtain, but its effectiveness as a fish barrier is greatly enhanced by the addition of a repellent sound signal. The characteristics of the sound signals are similar to those used in SPA systems, i.e. within the 20-500 Hz frequency range and using frequency or amplitude sweeps. Typically, the BAFF™ is used to divert fish from a major flow, e.g. entering a turbine, into the minor flow of a bywash channel. Recently, the Illinois Natural History Survey have conducted trials of the BAFF™ in a concrete raceway to assess its effectiveness as a barrier to the migration of invasive Asian carp species *Hypophthalmichthys nobilis* (Taylor *et al.*, in press). Initial trials using a 20-500Hz signal yielded only moderate performance, with 56% of approaches being successfully repelled. The signal was subsequently replaced by a 20-2000Hz signal, which increased deflection efficiency to 95%. This was comparable to results obtained with the GFFB electrical barrier in parallel tests.

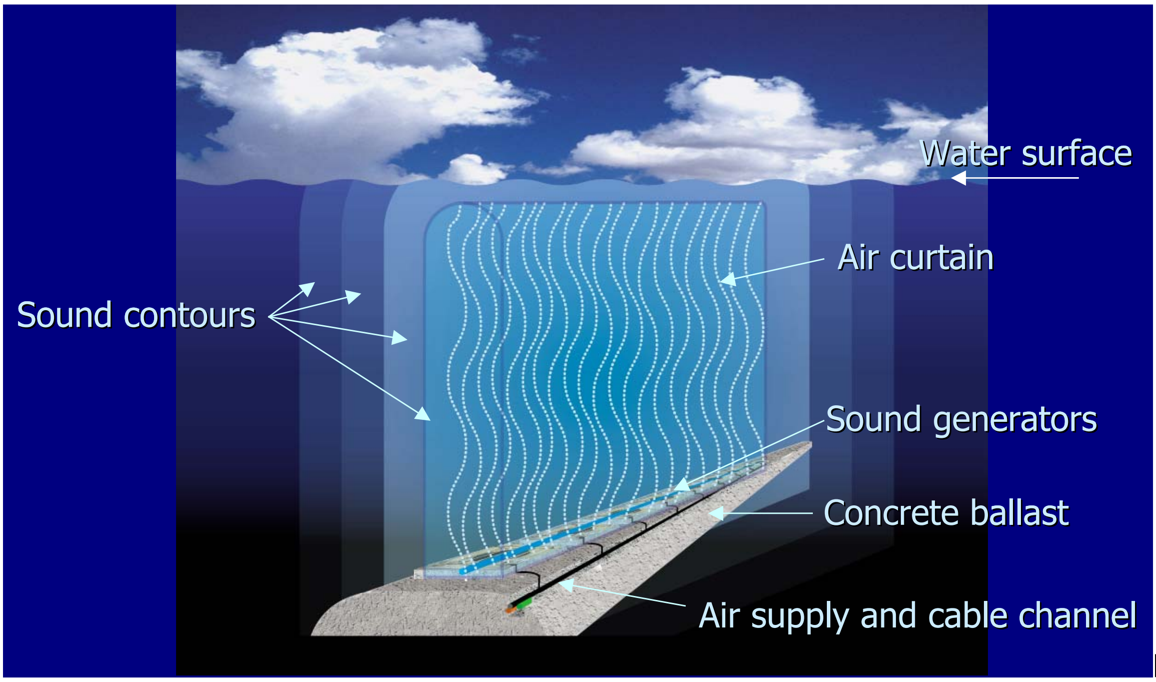


Figure 3.18 Schematic of BAFF™ acoustic bubble curtain (Fish Guidance Systems Ltd)

3.4.5.9 Installations in the UK

Investigations of a BAFF™ angled at 15^{o} system across a small (~5m width) mill stream of the River Frome (Dorset) yielded deflection rate of 20.3-43.8% in daylight and 72.9-73.8% in darkness with Atlantic salmon smolts (Welton *et al.*, 2002). The sound is generated pneumatically, with frequencies in the 50-600Hz band. A larger (24 m-long) BAFF placed at an angle across the main river to divert descending smolts into the mill stream for census purposes has regularly achieved efficiencies of 95-98% (S. Welton, personal communication). The river depth along the BAFF line is about 1.2m. The trials were conducted as part of an Environment Agency research programme. The better performance of the larger BAFF may have been due to the larger 'bywash' created by the entrance to the millstream. Observations at the BAFF located in the millstream demonstrated that many of the fish were effectively diverted by the BAFF but then turned back at the bywash entrance, owing perhaps to inadequate attraction flow. Such fish would often make several attempts, circling in the area upstream of the BAFF and finally 'rushing' the bubble curtain and passing through. A more detailed analysis of fish behaviour in front of a BAFF is provided by Turnpenny *et al*. (2002, in press), based on observations of fish fitted with float-tags.

A pneumatic BAFF is installed at a small hydropower scheme at Backbarrow on the R. Leven (Cumbria). It was installed in the headrace canal, principally to divert salmonid smolts into a bywash, which uses between 2 and 5% of the turbine flow as attraction flow. It also operates with frequencies in the 50-600Hz band. Performance trials have been carried out by the Environment Agency, by placing a rotary-screw smolt trap in the flow behind the BAFF to sample fish passing through the air curtain and simultaneously counting fish entering the bywash (Spiby, 2004). Bywash monitoring was carried out using a submerged video camera in the bywash entrance with infra-red illumination, connected to a video recorder. Observations were made over 44 days during the spring of 2003, during which time 109 fish were recorded. These were distributed as follows:

- 56.9% entered the bywash (video)
- 30.3% swam back into the headrace (video)
- 5.5% uncertain- either entered bywash or swam back into headrace (video)
- 7.3% passed through BAFF (rotary screw trap).

Spiby proposed a best estimate of 92.7% of fish being prevented from entering the turbine, although as only 56.9% were seen to enter the bywash, the true deflection efficiency may have been between 56.9% and 92.7%. Only 2.7% of the 7.3% of fish estimated to pass through the turbine were smolts. The author drew attention to a number of limitations of the study, particularly with regard to the quality of video monitoring and the performance of the trap under low flow or heavy weed conditions and recommended that further proving trials should be conducted.

3.4.5.10 Infrasound

Whereas the acoustic techniques described above may contain frequencies extending down into the infrasound (<20Hz) region, true infrasound devices are designed to emit primarily in this waveband. A review is given by Sand *et al.* (2001). Normally, the sound is generated by a mechanical, motor-driven device, driving pistons to generate high particle velocities in the region of the source. For Atlantic salmon smolts, sound intensities above $10^{-1}ms^{-2}$ at 10Hz are an effective deterrent and have been used successfully to block channels. Sand *et al*. mention mechanical reliability and metal fatigue problems with the source devices that have limited their practicability in the past but these may be reduced eliminated with further development. A particular interest with infrasound lies in the finding that adult silver eels (*Anguilla anguilla*) migrations were successfully influenced by an infrasound source in river trials. Audiogram measurements have shown that eels are most sensitive to sound pressure at frequencies centering on 90Hz but to vibrations of around 40Hz (Jerkø *et al*, 1989). Given the relatively limited range of screening methods suitable for eels, particularly in the hydropower and thermal power context, infrasound or low frequency sound merits further investigation.

3.4.5.11 Ultrasound Transducer Arrays

Ultrasound systems have so far been used mainly in north America, where arrays of ultrasound transmitters have been fitted around intake structures to repel shad and herring species (Carlson, 1995). Ultrasound may be worth considering for some UK applications, for example where shad are present, although shad also show a good sensitivity to LF systems, to which clupeids are more sensitive than ultrasound (Mann *et al*., 1997). The latter have the advantage of also repelling non-clupeid species.

3.4.5.12 Acoustic Attraction

While most studies have reported the use of acoustic stimuli as a fish deterrent, Patrick *et al.* (2001) demonstrated in tank experiments that eels (*Anguilla rostrata*) were attracted towards a transducer emitting a “complex signal” containing frequencies of <1000Hz. The signal characteristics were not described beyond this and the levels involved are not stated, although the source level was < 190µPa @1m. The authors suggest that sound may have some potential for attracting eels towards bywashes.

3.4.5.13 Design and Operational Best Practice

AFD systems should be installed according to manufacturers’ specifications. Important considerations are:

- Background noise levels should be measured prior to AFD specification to ensure that the signal is not going to be masked by noise from pumps, turbines, etc..
- For SPA systems, acoustic modelling (e.g. using PrISM™) is essential for all but the smallest applications. This should also be used to ensure that the spread of the sound field is not excessive, which might interfere with movements of migratory fish or cause local loss of habitat. Levels should also be measured at commissioning to validate predicted values.
- Diagnostic/monitoring systems should be fitted so that the performance of the underwater equipment can be monitored e.g. from a plant control room. Some form of indicator should be fitted at the abstraction point to show the operational status to operational and enforcement staff.
- Provision should be made for retrieving the underwater equipment for servicing.
- Some redundancy (i.e. using more sound sources than are strictly needed) is desirable to allow for sound projector failures.
- The equipment requires regular maintenance and service intervals should be displayed and logged in the plant control room.
- Back-up power or interlocks with pump controls may need to be provided to ensure that pumping does not occur when the system has lost power.

- In the case of the BAFF™, the recommendations made regarding bubble curtains (see above) also apply, as do physical limitations regarding water depth and flow.

3.4.5.14 Applications

Acoustic fish deterrents are particularly suited to high flow rate intakes where positive exclusion screens are impractical owing to the hydraulic head loss or the risk of screen blockage and where <100% exclusion is acceptable. Electrical power is required, which makes them unsuitable for unpowered, remote sites. There are more than sixty installations in Britain, Europe and North America, with applications including potable water intakes, flood relief pumping stations, thermal and hydroelectric power plant and a fish census station. SPA and BAFF™ are also being tested in the USA for possible use as invasive species barriers against the spread of Asian carp species, with excellent results to date (Taylor *et al*., in press).

Since estuarine and coastal power stations tend to draw in large quantities of hearing-sensitive pelagic species, such as sprat, herring and smelt, AFDs have been found to yield large reductions in catch of these species. Systems have recently been installed in the UK at new CCGT power stations at Shoreham (W. Sussex) and Great Yarmouth (Norfolk) and retrofitted to Fawley Power Station (Hampshire). These stations also operate fish return techniques that put back to the wild any hearing-insensitive animals that get past the AFD system. These are mainly eels, flatfishes and other benthic species, as well as shrimps and crabs. Post-commissioning surveys conducted at Shoreham in the three years since its completion suggest that it has the lowest fish catch of any major British coastal power station.

3.4.5.15 Species and Lifestages

AFD systems are best suited to fishes with moderate to high hearing sensitivity. Best results are obtained with hearing specialists, e.g. clupeids (herrings and shads), cyprinids, other sensitive species including smelt (*Osmerus eperlanus*) and bass (*Dicentrarchus labrax*). Non-specialist species with a fully developed swim bladder are also amenable to AFD guidance, e.g. salmonids and gadoids. Species with a poorly developed or no swim bladder, e.g. most benthic species, such as flatfish, can be deterred using high sound levels but this may not be cost-effective. In particular, eels and lampreys show very little reaction to AFD signals, although the use of infrasound merits further research.

3.4.5.16 Ease of Retrofitting

AFD systems lend themselves to retrofitting, requiring very little engineering work to install. As well as being used to upgrade fish protection on older intakes, for example by Thames Water Utilities Ltd on R. Thames intakes, Yorkshire Water plc have used them for fish diversion at temporary water abstractions.

3.4.6 Light-based Systems

Light is used in two ways to reduce entrapment. The first is to illuminate physical or behavioural screens to make them more visible so that fish can orientate themselves in relation to the flow (using the optomotor response); the second is to use the stimulus of light in its own right to either attract or repel.

The fish deterrent effects of light were first studied in the 1950s when Brett and MacKinnon (1953) used light to restrict the movement of animals in a canal. Although these early tests were not extensive there were two important findings: firstly, that different reactions were displayed by different species and secondly that flashing lights were more effective at eliciting a response than continuous light (OTA, 1995).

It is commonly found when using physical screens, greatest impingement occurs during hours of darkness. Up to 97% of young fish will be entrained at night (Pavlov, 1989). During hours of darkness in particular, artificial lighting will allow improved orientation of fish and therefore reduced entrainment. This effect may be further enhanced by careful positioning of light sources behind structural elements to provide maximum visual contrast by throwing the structure into silhouette (Turnpenny, 1998). With the addition of lights to an intake structure Pavlov achieved a reduction of entrainment of young cyprinids and percids of up to 91%. The effectiveness of this behavioural system, however, varies with species with up to 100% deterrence being seen for perch (*Perca fluviatilis*) and ruffe (*Gymnocephalus cernua*) whereas entrapment was increased with illumination in the case of three-spined-sticklebacks (*Gasterosteus aculeatus*) (Hadderingh, 1982). Light can therefore act as either a repellent or an attractant to different species.

To minimise light pollution and to achieve the highest possible effectiveness it is necessary to submerge the light source. This results in a significant increase in the capital and maintenance cost due to necessary frequent cleaning of lamps requiring a mechanical recovery system. The most common positioning of lamps is in an arc on the bed around the intake entrance ensuring water velocities at this point are low enough to allow fish to escape (Turnpenny, 1998). The angle of positioning of the lamps is also important with an upwards tilt of 40-45° having been found to be most effective (Johnson *et al*., 2001).

3.4.6.1 Constant Illumination

Continuous illumination is not the optimum method for most species but is useful in the case of eels. This approach has been tested extensively in the Netherlands (Hadderingh and Smythe, 1997). Eels show strong phototaxis and positive rheotaxis (orientation into currents). Light can therefore be used to discourage the tendency of eels to follow water flow. The lights can be incandescent lights, mercury vapour lights or fluorescent lights. Trials have mainly used the latter (specified as 36W, PL-L Philips, spectrum with peaks at 440, 550 and 610nm). Deflection rates of up to 74% have been observed at some thermal and hydroelectric power stations.

3.4.6.2 Strobe lights

Strobe lights generally give better results than continuous illumination. Most experiments have again centred on the eel. Patrick *et al.* (1982, 2001) conducted experiments on eels. The first study was to determine if strobe lights could be used to deter eels from entering a turbine unit during its shutdown period. The second involved initial laboratory tests followed by field trials at a fish ladder. Both investigations showed a reduction of eel movement of between 65% and 92%. The laboratory tests in the second study used flash frequencies from 66 to 1090 flashes per minute (FPM) and showed that all were effecting in repelling eels. The fish ladder trials used a flash rate of >800 FPM. The threshold light level for eel repulsion was found to be $\geq 0.1 \mu Em^{-2}s^{-1}$ ($\geq$5 lux).

Work has also been carried out on other species, including white perch (*Morone americana*), spot (*Leiostomus xanthurus*) and Atlantic menhaden (*Brevoortia tyrannus*) although the level of effectiveness and the necessary flash rate varied with species (Sager *et al*., 1987). Other species investigated by Patrick *et al.*(1982) included Atlantic salmon, which were also repelled by strobe lights. Flash rates of 300 FPM appeared to be most effective. Johnson *et al* (2001) also found 300 FPM to be effective in achieving vertical displacement of steelhead trout (*Oncorhynchus mykiss*); their concept was to shift the fish upwards in the water column away from an intake to reduce entrainment. Tests with kokanee (*O. nerka*) found flash rates of 300, 360 and 450 FPM all to be an effective repellent, even when light levels were <0.00016 lux above ambient light levels; the repellent effect continued throughout the longest test duration of 5h 50min (Maiolie *et al*, 2001).

The success of using strobe lights as a deterrent has been found to be site specific, indicating that hydraulic and environmental conditions have an effect (OTA, 1995).

Strobe lights may prove to be more effective when used in conjunction with other forms of behavioural and physical screening systems. In particular bubble screens /strobe light combinations work well for some species, e.g. alewife (*Alosa pseudoharengus*), smelt (*Osmerus mordax*) and gizzard shad (*Dorosoma cepedianum*) (Patrick *et al*., 1985). This combination was also tested in the UK at Walton-on-Thames raw water intake, where a reduction of 62.5% entrainment of salmon smolts was observed (Solomon, 1992).

In earlier systems a problem with the short lifespan of the xenon discharge tube made operation difficult. Modern tubes however, will last for a year or more when correctly driven. New models of strobe lights are now available with greater flexibility in flash rate, light intensity and sequencing via a laptop computer. This will allow for easier adjustment of the lights without the need for removal, therefore reducing costs (Johnson *et al*., 2001).

3.4.6.3 Design and Operational Best Practice

There has been very little use of this approach in the UK to date and therefore 'best practice' is unclear. Important issues are:

- Water clarity must be high.
- A lamp retrieval mechanism must be installed.
- Adequate testing is required to optimize the flash rate when strobes are used; there is a risk of attracting rather than repelling fish at some flash rates. Flash rates of ≥300 FPM appear to work best with a range of species.
- Some redundancy (i.e. using more lights than are strictly needed) is desirable to allow for lamp failures.
- Warning systems (e.g. via telemetry links) should be provided to inform plant operators of failure.
- The equipment requires regular maintenance and service intervals should be displayed and logged in the plant control room.
- Visible indicators of the operational status (e.g. number lamps operating versus failures) should be displayed at or close to the intake to inform operational and enforcement personnel.
- Back-up power or interlocks with pump controls may need to be provided to ensure that pumping does not occur when the system has lost power.

The use of 'high-tech' computer control systems appears to enhance flexibility and control of the systems and looks promising.

3.4.6.4 Applications

Light-based techniques are appropriate in similar situations to acoustic methods, i.e. where large flows are to be screened with zero headloss (e.g. hydroelectric and thermal power plant intakes). However, unlike acoustic methods they are not suitable for turbid waters.

Strobe light systems appear to work well in combination with bubble curtains from the limited research available. A BAFF™/strobe combination in particular merits investigation. Recent improvements in strobe lamp technology, which now offer greatly extended operating life, make the technology potentially more useful.

3.4.6.5 Fish Species/Lifestages

Light-based methods show promise for eel guidance in particular, although a number of other species can be deterred using strobe lights. Combinations of AFDs and strobe lights are worth considering where eels need to be deterred along with acoustically sensitive species.

3.4.6.6 Ease of Retrofitting

Light-based systems of any kind are relatively easy to install and provide an attractive retrofit option.

3.4.7 Velocity Caps and Other Flow Control Measures for Offshore Intakes

The velocity cap represents a simple modification to unscreened intakes in open sea or lake situations which can significantly reduce entrainment. Many coastal power stations constructed before the 1970s used vertically opening offshore “bath-plug” intakes which are prone to draw fish down (Schuler and Larson, 1975; Hocutt and Edinger, 1980). This occurs because fish are adapted to respond to horizontal rather than vertical currents. The velocity cap, usually made of concrete or steel, forms a flat, horizontal lid to the intake that therefore draws water in horizontally. Schuler and Larson (1975) proposed that the cap and lip of the intake riser should extend out 1.5 times the height of the intake opening (Figure 3.19). This straightens the flow and allows the fish some distance over which to react. These authors reported substantial reductions in pelagic fish entrainment when a velocity cap was fitted to a Californian power plant intake. Velocity capping of some form has now become standard practice in offshore cooling water intake design, as it also offers the advantage of selective withdrawal of water from the cooler, deeper layers (Turnpenny, 1988).

Turnpenny (1988) identified a further fish entrainment issue associated with offshore intakes drawing from tidal streams. Whereas in still water, velocities will be radially symmetrical around a circular intake, in flowing water the water is abstracted primarily from the upstream side, giving rise to higher intake velocities on this side. At Sizewell ‘A’ nuclear power station (Suffolk), it was shown that fish impingement on the drum screens peaked on the mid-ebb and mid-flood tides when this effect became maximal. Physical and numerical hydraulic modelling studies (Turnpenny 1988 and unpublished) have shown that blanking off the upstream and downstream sides of an intake, as well as velocity capping it, can provide an intake velocity regime that remains consistently favourable for fish throughout the tidal cycle. A suitable design which has emerged from the model tests uses two circular caissons placed in line with the intake, with a velocity cap extending across the whole structure; water is abstracted from a central seabed port (Figure 3.20). At present, this design has not been built at full-scale.

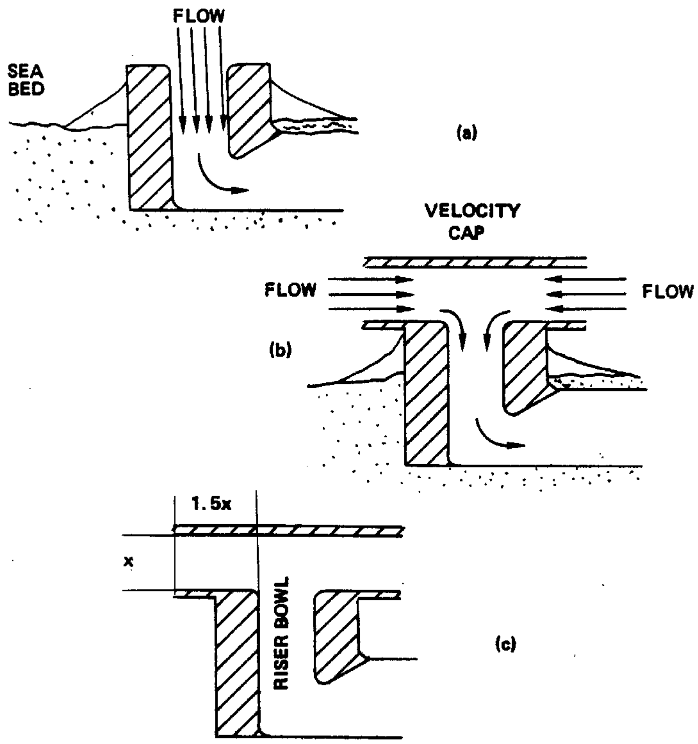


Figure 3.19 The velocity cap: (a) section of uncapped intake showing vertical draw-down pattern, (b) section of capped intake showing horizontal flow pattern, (c) as (b) but showing critical relationship between vertical opening [x] and length of horizontal entrance [1.5x] for fish reactions (after Schuler and Larson, 1975).

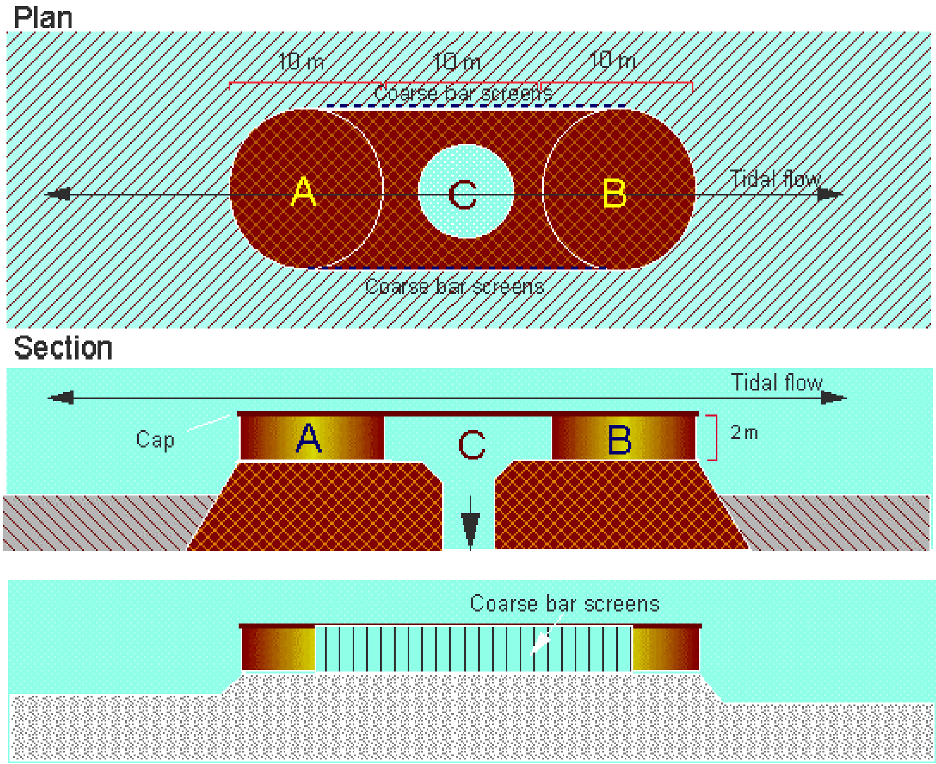


Figure 3.20 Concept design for a low-velocity, side-entry offshore intake structure with velocity cap, based on physical and hydraulic model tests carried out at Fawley Aquatic Research laboratories (Turnpenny, 1988 and unpublished). A and B are the cylindrical caissons. Water enters C.

3.4.7.1 Design and Operational Best Practice

- Where it is not feasible to use more effective fish screening methods such as PWWC screens, offshore intakes should be designed to ensure that flows are kept horizontal and that intake velocities are not unduly influenced by tidal movement or river flow. These conditions are best achieved by fitting a velocity cap and lateral intake ports.
- Offshore intakes are usually costly structures and it is strongly recommended that physical or numerical hydraulic modelling studies are undertaken to demonstrate that the above intake flow conditions are attained.
- Experience at UK sites where velocity caps are present has shown that fish entrainment remains a problem and velocity caps are therefore not in themselves a solution. Other measures, including use of side-entry, of behavioural deterrents and onshore fish return systems need to be considered.

3.4.7.2 Applications

The measures described should be applied to all offshore intakes and mid-river intakes where fish are not protected by more stringent screening measures such as PWWC screens.

3.4.7.3 Fish Species/Lifestages

Velocity caps are most important for the protection of pelagic species or lifestages that are found in the upper water column. At most power plant intakes, pelagic fish are usually overwhelmingly the major entrained component. Side-entry devices are appropriate to all species and lifestages.

3.4.7.4 Installations in the UK

Many UK offshore intakes have capped intake structures but few can be described as true velocity caps as described by Schuler and Larson (1975). Dungeness 'B' (Kent) and Sizewell 'B' (Suffolk) nuclear power stations both have velocity caps, as does Stallingborough power station (Humber estuary). None are known to have any device deliberately to control the radial distribution of intake flow for the purpose of reducing entrainment.

3.4.7.5 Ease of Retrofitting

Some attempts to reduce vertical draw have been made on offshore intakes. At Dungeness 'A' power station, the top bars of the original caged structure were blanked off with concrete to reduce vertical flow, although this does not form a true velocity cap. A badly designed intake (from the fish protection point of view) cannot easily be remedied without major civil works. Acoustic fish deterrents and fish return systems have so far proved the best remediation option for such cases.

3.5 Other Behavioural Guidance Techniques

There are examples of other interesting techniques that have been investigated in North America and which may have some potential for use in Britain.

3.5.1 Turbulent Attraction Flow

The *turbulent attraction flow* concept attempts to mimic natural river turbulence cues that downstream migrating fish use to follow currents (Coutant, 2001). Any stream or river has turbulences along its watercourse due to interactions with the topographic features of the riverbed. These turbulences alter the velocity, direction and pressure of the water flow. One main form of turbulence found in a river is a 'turbulent burst', which is a high-speed ejection of water and suspended solids as water passes an obstruction. Rows of vortices are another common feature created in the wake behind solid objects. It is believed that migrating juvenile salmon use the enhanced water velocities found within these turbulences to assist in their downstream migration. While high turbulence intensities and small vortex diameters can be damaging to fish, low intensities create an attractive stimulus.

It follows from this that low turbulence intensities can be created to attract fish to a bywash entrance. A trail of turbulence can be engineered either actively or passively. Passive devices are structures such as concrete cylinders that are placed on the riverbed. When placed at intervals, a chain of vortices and increasing velocity can be generated, leading into the bywash. Where there is insufficient momentum in the water, active devices may be necessary. Turbulence can be actively created using devices such as a pumped water jet, propeller or paddle wheel suspended midway in the water column (Coutant, 2001).

A prototype 'current inducer' was tested by Truebe *et al*. (1997, 1998). A mechanically generated current was created to direct Atlantic salmon to a surface bywash system.

Prior to installation of the device, natural flow was minimal within the area and fish were drawn towards the turbine intake. The current was generated by two 2-hp, low-speed electrically driven propellers. The system resulted in a bypass efficiency of up to 93% (Coutant, 2001).

Turbulence induction may be most useful as an adjunct to other behavioural and positive exclusion screening technologies. This area deserves further research to establish design criteria and its effectiveness with different fish species and lifestages.

3.5.2 Surface Collectors

This fish diversion technique, used at large dams, is based upon the natural tendency of salmonid smolts to migrate in the surface layers of the water column, allowing fish to be skimmed off by surface bypasses or 'collectors'. The surface collectors are located on the dam and fish can be either bypassed around the dam or transported downstream in trucks or barges.

Until very recently, surface collectors have been implemented in the field only as prototypes. Prototype trials have been carried out by the US Army Corps of Engineers in the Pacific Northwest region of the USA and are the source for the following information (Dankel, 1999; Lemon, 2000).

One of the largest of these prototypes was installed at the Bonneville Dam on the Columbia River, Oregon in 1996. The main structure of the dam houses ten Kaplan turbine. The surface collector has 12 modules spanning four of the ten turbine bays. The modules are 21 m high, 8.4 m wide and 7 m deep. An attraction flow is provided by means of current inducers located within each unit. The system then diverts fish towards an existing fish screen, which channels them around the dam. The surface collector system is expected to divert 50-60% of approaching fish whereas the combination of the surface collector and the existing screen system should achieve 90% diversion.

In 2001 a seven-year prototype trial surface collector screen was removed from the Rocky Reach hydroelectric project on the Columbia River (Plates 3.14, 3.15). It has been replaced with one of the first permanent systems, which began full operation in April 2003 (www.chelanpud.org)[20]. The system is made up of two parts. First is the collector system itself which uses 29 pumps to create a strong current in the upper 60 feet of the flow to attract fish away from the turbine flow into the bypass system. The bypass system consists of a 7.7 m diameter tube which passes through the dam and extends some 1,380 m around the back of the powerhouse and 0.5 km down the east bank of the river. The total trip for fish passing through the system is 6 to 8 minutes long. The system cost $112 million to construct; this is expected to be recovered by the $400 million saving within 15 years due to a reduction in spill loss. The system is expected to achieve a 98% fish survival rate.

[20] Chelan County Public Utilities District – Rocky Reach Hydro Project

Plate 3.14 Rocky Reach dam with the powerhouse on the left and the bypass pipe in the bottom right (Chelan County PUD).

Plate 3.15 The bypass pipe traversing the dam (Chelan County PUD).

The system is still being monitored in the field and needs further investigation both in the field and the laboratory before widespread installation.

3.5.3 Eel Bypasses

One further technique of interest is the eel bypass channel. An example is found at Backbarrow hydropower scheme on the R. Leven in Cumbria (Spiby, 2004), which is claimed to be based on a traditional eel fishing method. The bypass comprises a trough set in the floor of the headrace at an angle of 60^{o} to the flow, with a 20 cm high wall on the downstream side (Plate 3.15) and a 20 cm bywash pipe at its downstream end. The effectiveness of this bypass is not known. Similar arrangements are reported in Richkus (2001), who cites French and German examples of this method used for European eel (*Anguilla anguilla*) diversion (Travade, unpublished and Rathke, 1993, unreferenced). The French example refers to the Halsou hydroelectric project in the Pyrenees, where a deep trough is set into the floor of the headrace upstream of the trash racks. This connects to a bywash, which draws 3-5% of the turbine flow. From radio-tracking studies, it was estimated that between 50% and 80% of eels used the deep bypass. In Germany, in one case the eel bypass was formed by a steel half-pipe set into the floor of a turbine spiral chamber. When fully opened, the pipe carried 1000 Ls^{-1} (proportion of total turbine flow unspecified). Using farmed (presumably yellow) eels, 41% of eels released used the bypass. No further information is given. Another German example from the same author used a bypass depression of 50 cm wide and 15 cm deep across the width of the intake channel, leading into a 25 cm bywash pipe. From studies carried out during silver eel migrations, this was reported to be used by “a high percentage” of eels at low and intermediate river discharge, but a smaller percentage at higher flows. Richkus (2001) provides a comprehensive review of downstream eel migration and deflection technologies, from which the evidence is that concerted silver eel migrations tend to occur on high river discharges. This would suggest that this technique is not enough by itself to protect eels. Nonetheless, given the relatively poor performance of virtually every other technique against eels, it is clearly a potentially valuable option.

Richkus (2001) concludes that to date there has been no rigorous research to either optimize or evaluate the eel bypass technique. Available information suggests that eel bypasses should have entrance velocities the same as those occurring at the intake trash racks and that bypass flows should be 3-5 % or more of total river discharge. Presumably in the case of a hydropower scheme or other lade-type offtake, the bypass flow should be 5% of the channel flow, not the whole river discharge; this would accord with the upper end of the range 2-5% of turbine flow commonly recommended for fish bypasses at hydropower sites (Turnpenny *et al*, 1998).

Plate 3.15 Eel bypass trough at Backbarrow hydropower scheme, R. Leven, Cumbria (Spiby, 2004).

3.6 Outfall Screening

3.6.1 Introduction

The screening of outfalls is for one or both of two purposes:

1. to prevent upstream swimming fishes that may be attracted to a discharge flow from entering the discharge or being distracted from the natural flow;
2. to avoid losses of fish stock from a fish farm, reservoir or pond outlet into a natural watercourse.

Outfall screens are limited to two types that have been shown to be effective: mechanical mesh or bar screens and electric screens or 'hecks'. Acoustic methods, and possibly other behavioural methods, have been attempted to block upstream movements, but largely unsuccessfully. The lack of reliably good results appears to be because of the strong motivation of fish as they migrate towards their spawning grounds: stimuli that might under other circumstances deter them become ineffective.

Screening against up-migrating elvers has not traditionally been practiced but must now be considered, in the light of the current declining status of eel stocks. This is probably not an issue at hydropower sites or other large, high velocity outfalls that elvers would find difficult to ascend, but may be at low velocity outfalls. Swimming speed measurements for elvers of *A. anguilla* made as part of the Environment Agency National R & D Project No. W2-049 "Swimming Speeds in Fish" indicate that upstream-migrating elvers can attain an average burst speed of 0.5 ms^{-1}, suggesting that the discharge velocity would need to be at least 0.7-0.8 ms^{-1} to ensure that elvers could not ascend. However, this in itself could attract larger migratory fish and therefore these would need to be screened out. There are options other than screening; for example, since eels cannot leap, raising the discharge point above flood water level would be effective.

3.6.2 Positive exclusion Screens

From the fish protection point of view, there are a number of criteria for effective outfall screening:

3.6.2.1 Mesh or Bar Spacing

The screen should have mesh or bar spacings suited to the sizes of fish to be excluded. Standard sizes are e.g. 40 mm horizontal spacings (free gap) for adult Atlantic salmon or 30 mm for adult sea trout (Anon., 1995). Smaller spacings may be required for other species (see section 3.1). Square or rectangular bar is preferable to round bar for fish screens.

Fish farm outfalls represent a special case, where screening needs to be provided to keep farm fish inside the farm without allowing escapes to the watercourse. In this case, the screening should reflect the sizes of fish held on the farm. Normally, however, this function is provided by fish screens located at strategic points around the farm, and not necessarily at the final outfall position. An outfall screen suited to the river fish will then need to be provided at the point of outfall.

3.6.2.2 Screen Location

Outfall screens should be located at the most downstream point of the discharge; failure to do this can create blind alleys where fish become trapped and possibly vulnerable to poachers. The position and alignment of the screen can be arranged to guide fish upstream towards the preferred route, e.g. the main river channel or the entrance to a fish pass. This is particularly important where screens are placed across hydroelectric tailraces.

3.6.2.3 Working Depth

Where screens are not e.g. fitted to cover the end of a pipe or tunnel but are placed across an open channel, the height and the extent of the screen should take account of the local topography and foreseeable flood levels; otherwise fish may circumvent the screens during floods and become trapped when the level falls. The same proviso applies to screens used to retain fish on fish farms.

3.6.3 *Electric Barriers*

Electric barriers or hecks have been used for many years to prevent the ascent of fish into hydroelectric tailraces and they are generally considered to work well for the purpose. This method is sometimes preferred to positive exclusion screening at low-head hydro sites, where the additional loss of head caused by the screens may be significant. Most of the MAFF-type electric screens (see section 3.3) have now, however, been removed owing to safety concerns.

The GFFB system described in section 3.3 is also used for outfall screening but is considered to be a safer option, for reasons discussed in section 3.3. The GFFB can be fitted as an electrode array running across the bed (Figure 3.21) or as an annular array fixed within the confines of a tunnel. It is found to work best with a minimum water velocity of 0.6-0.9 $m.s^{-1}$, which causes fish to be pushed away as they cross the electric field, and with a maximum operating depth of about 5 m.

In an investigation carried out at the Pere Marquette River (Rozich, 1989) the GFFB system was found to be effective in preventing upstream sea lamprey migration whilst still allowing downstream penetration of steelhead adults and smolts and chinook smolts with no injury.

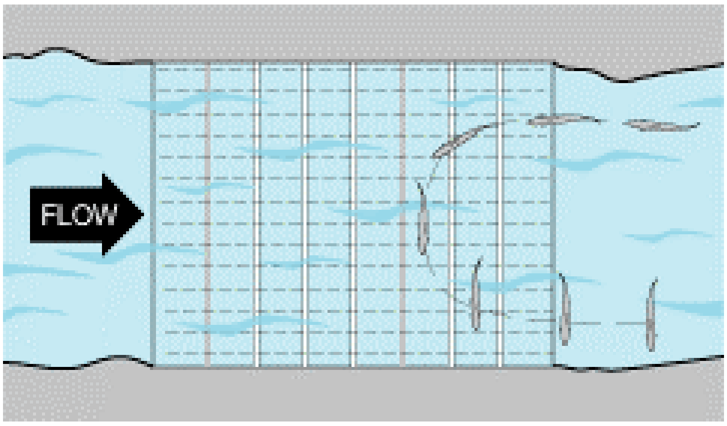


Figure 3.21 Diagram showing fish movement in response to the GFFB electric fish barrier. In the diagram, the energisation voltage of the electrode strips increases from right to left of the diagram. The fish turn away at a point depending on their length and hence the voltage received along the body (www.smith-root.com).

Two GFFB™ barriers have recently been installed in Britain for deflecting upstream migrants. One, fitted into the draft tubes of a small hydroelectric plant at Beeston Weir (R. Trent, Notts.), has been in operation for about five years. It is intended to prevent the ascent of salmon and adult coarse fish into the turbines rather than the adjacent vertical-slot fish pass. While no formal efficiency testing has been undertaken, it appears to have worked well when operated correctly (see comment in section 2.2).

3.6.3.1 Design and Operational Best Practice

Key points are:

- All fish farm outfalls should be screened to prevent accidental loss of fish from the site at the 10-year return period flood level.
- Other outfalls should be screened or raised above flood level to prevent the risk of fish ascending the discharge.
- In the case of physical screens, mesh sizes or bar-spacings should be selected based on data given in section 5.7.
- For electric screens, the electric field strength should be set as per the manufacturer's recommendations for the fish species, sizes and site conditions. Manufacturer's health and safety guidance should also be followed. For public safety, it is desirable to limit access to electrified areas, and to display adequate warning notices. Other relevant data are given in section 3.4.4.
- Screens on tailraces and other types of outfall channel should be located at the confluence with the natural channel and not at the upstream end of a blind channel.
- Outfall screens on fish farms should be constructed sufficiently sturdily to withstand flood conditions.

4 PERFORMANCE CRITERIA

4.1 How Effective Should a Fish Screen Be?

There is a common misconception that all positive exclusion fish screens, provided that they are designed with the optimum mesh size and velocity conditions are 100% effective. In practice, this success rate is seldom achieved. Inspection surveys frequently reveal faults in the operation or maintenance of even the best designed screening systems. Common faults with mesh panel screens for example include:

- Damaged mesh panels;
- Damaged screen seals;
- Screens not fully seated;
- Screens removed to avoid clogging problems;
- Screens heavily clogged, leading velocity hot-spots where fish are at risk of becoming impinged on the screens.

These, of course, can all be overcome with appropriate monitoring, maintenance and enforcement.

Certain types of positive exclusion screen are much less prone to maintenance failures like these, for example PWWC screens or rotary disc screens are inherently proof against fish entry, unless they become seriously damaged by flood debris. Coanda screens also offer a high degree of protection, provided that they are operated with sufficient surplus flow to allow fish to pass. Where the sensitive status of the fishery demands near-100% efficiency, these methods should be used, if feasible.

Physical and cost constraints of particular sites, environments or applications dictate that these methods are not always viable, which is why a variety of methods has grown up to provide alternative solutions. Behavioural methods have been developed primarily to deal with issues of high waterborne detritus loads, and risks of screen clogging and consequent hydraulic losses. These issues are particularly critical to hydroelectric generation, where flow and operating head equate directly to revenue, but also to e.g. thermal power generation and other industries where loss of the water supply might be critical to operation or safety. For large coastal power stations, no solution has been found to date that will yield near to 100% fish exclusion. The problems include: very high biofouling rates of submerged screens, inundation by weed, jellyfish, shrimps, crabs and even sprat shoals and very high rates of water abstraction (60 m^3s^{-1} for a 2000 MWe fossil fuel plant: Turnpenny and Coughlan, 2003).

Meeting the conflicting demands of an industrial society and the need for ecological conservation means that it is therefore essential to establish what the performance criteria for screening system should be before selecting the screening method.

4.2 Risk Assessment for Fish Screening

4.2.1 In General

The risk assessment approach requires that the effectiveness of the screening measures should reflect the level of risk to the fish stock or fish community and the importance attached to the stock, community or associated habitat. It is strongly recommended that the principles of risk analysis are applied to any intake screening proposal. However, the requirements of the relevant legislation must be taken fully into account.

Risk assessment in the fish screening context was addressed by Turnpenny *et al.*, (1998, 2000), who identified a number of factors that may be used in a risk assessment of an abstraction scheme (hydroelectric schemes in their examples, but equally applicable to other types):

- the value of the fish stock in economic or conservation terms;
- the percentage of the fish stock that must pass the scheme;
- the percentage of those fish that pass successfully;
- the additional loss due to other schemes (i.e. cumulative impacts);
- the significance of given percentage levels of loss in economic and conservation terms.

These authors present simple mathematical approaches that could be extended to different applications. For example, they demonstrated that a behavioural barrier having a 90% efficiency might achieve a successful scheme bypass rate of 98% when all factors (proportion of flow screened, etc.) are taken into account.

4.2.2 For Hydropower Sites

Specific consideration should be given to the following additional aspects at hydropower sites:

- the risk of fish injuries or mortalities in the turbines (via both intakes and outfalls);
- possible delays in fish migration and increased predation risk when water is diverted through long head- and tailrace systems;
- possible losses at bywash outfalls where the increased concentration of diverted fish may attract predators, especially if fish are disorientated.

Issues associated with fish passage through turbines have been investigated in some detail in studies funded by the Department of Trade and Industry (Turnpenny, 1998; Turnpenny *et al.*, 1998, 2000). These authors conducted a series of laboratory and field experiments, culminating in a computer model ('STRIKER') which predicts the probable injury rates of fish of different sizes during passage through a hydroelectric turbine of the Francis or Kaplan/propeller types. The model was validated on operating turbine sites. It takes account of injuries due to the following effects:

1. Runner strike- i.e. contact with the turbine blade;
2. Contact with fixed elements (guide vanes etc.);
3. Hydraulic shear stress (caused by hydraulic anomalies near the moving blades);
4. Rapid pressure change (as the fish passes from the high-pressure to low-pressure side of the blade).

Turnpenny *et al.* (1998, 2000) describe how the model can be applied in fish risk assessment for a hydropower scheme.

Methods of examining some of these factors on operating sites are discussed in section 7.

4.2.3 For Water Transfer Schemes

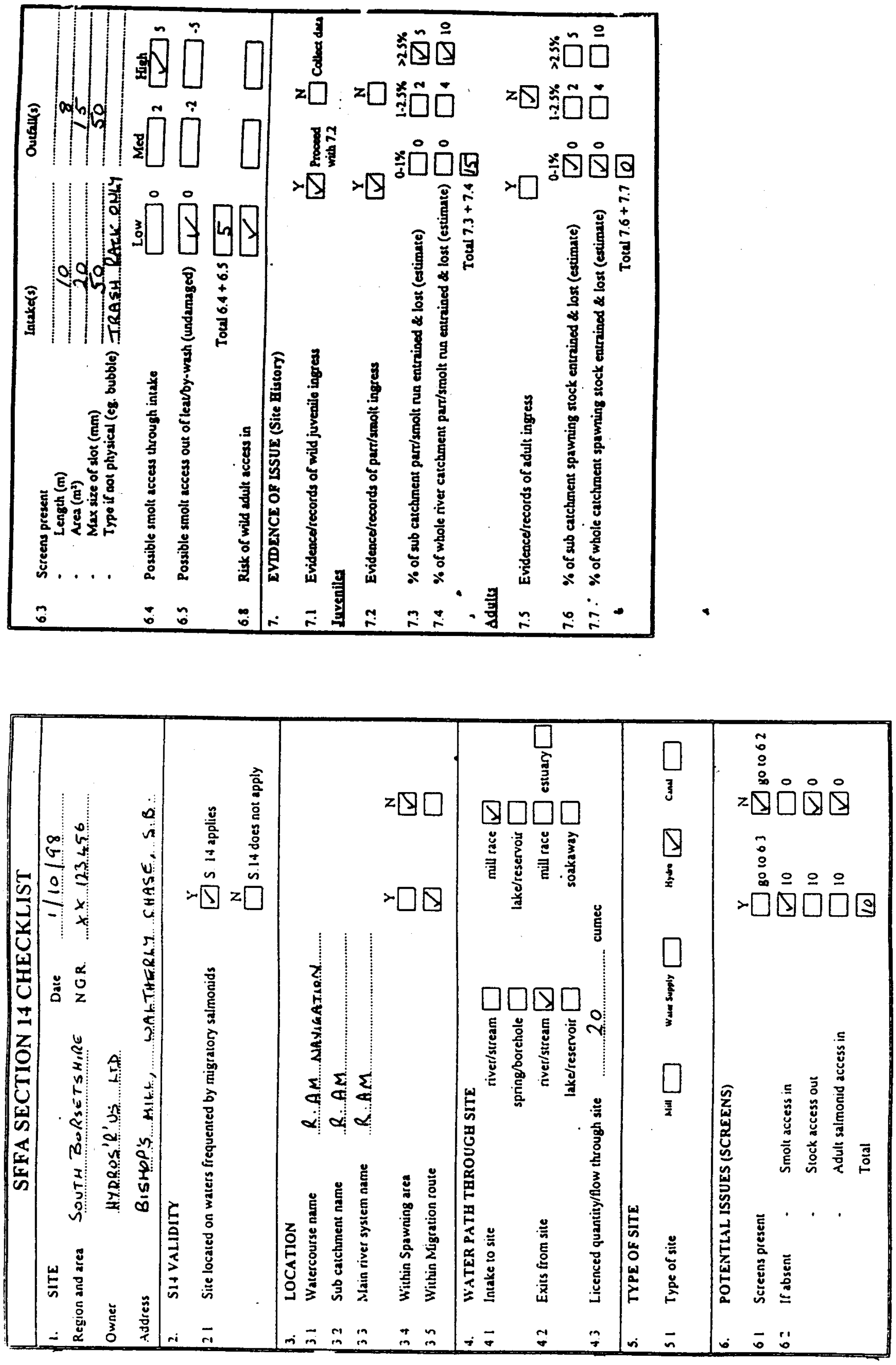
Risk assessment for procedures for screening at water transfer schemes, whether local (e.g. canal lock recharge pumping) or long-distance, should take account not only of impacts on fish communities in the source water body but also in the receiving water. This should include consideration of possible transfer of non-indigenous species as eggs, fry or other stages, spread of disease and interference with genetic integrity.

4.2.4 Under SFFA s.14

Another approach is exemplified by the Environment Agency's SFFA *s*. 14 risk assessment procedure. The Environment Agency is keen to demonstrate an effective but transparent implementation of the SFFA *s*.14. The key elements of the Environment Agency's *s*.14 policy are:

1. A standard risk assessment checklist procedure is used, the results of which are made available to the responsible person/ owner and open to appeal.
2. Full recognition is given to the precautionary approach ("where there are threats of serious or irreversible damage, lack of full scientific certainty shall not be used as a reason for postponing cost-effective measures to prevent environmental degradation"). This is complementary to the approach adopted in the Habitats Regulations.
3. The Agency will ensure that adequate screening provisions are implemented but it is recognised that the *costs to industry should be in proportion to the magnitude of the perceived risk*. At all times the principle of 'best available technology not entailing excessive cost' (BATNEEC[21]) should be employed in options for screening.
4. Site inspections may only be carried out by enforcement staff who have received formal training in the new SFFA *s*.14 procedure. Such staff are required not to offer advice on the design and construction of the suitable screening arrangements, which are the responsibility of the responsible person or owner.
5. Figure 4.1 shows an example of the Agency's SFFA *s*.14 checklist used as a basis for the risk assessment.

[21] Note that there is no formal definition of Best Available Technology (*sensu* Environment Act 1995) for fish screening at present



SFFA SECTION 14 CHECKLIST

1. SITE — Date: 1/10/98
Region and area: SOUTH BORSETSHIRE — NGR: XX 123456
Owner: HYDROS 'R' US LTD
Address: BISHOP'S MILL, WALTHERLY CHASE, S.B.

2. S14 VALIDITY
2.1 Site located on waters frequented by migratory salmonids — Y [✓] S.14 applies; N [ ] S.14 does not apply

3. LOCATION
3.1 Watercourse name: R. AM NAVIGATION
3.2 Sub catchment name: R. AM
3.3 Main river system name: R. AM
3.4 Within Spawning area — Y [ ] N [✓]
3.5 Within Migration route — Y [✓] N [ ]

4. WATER PATH THROUGH SITE
4.1 Intake to site — river/stream [ ] spring/borehole [ ] mill race [✓] lake/reservoir [ ]
4.2 Exits from site — river/stream [✓] lake/reservoir [ ] mill race [ ] soakaway [ ] estuary [ ]
4.3 Licenced quantity/flow through site: 20 cumec

5. TYPE OF SITE
5.1 Type of site — Mill [ ] Water Supply [ ] Hydro [✓] Canal [ ]

6. POTENTIAL ISSUES (SCREENS)
6.1 Screens present — Y [ ] go to 6.3; N [✓] go to 6.2
6.2 If absent - Smolt access in — Y [✓] 10; N [ ] 0
- Stock access out — Y [ ] 10; N [✓] 0
- Adult salmonid access in — Y [ ] 10; N [✓] 0
Total [10]

		Intake(s)	Outfall(s)
6.3	Screens present		
	- Length (m)	10	8
	- Area (m²)	20	15
	- Max size of slot (mm)	50	50
	- Type if not physical (eg. bubble)	TRASH RACK ONLY	

6.4 Possible smolt access through intake — Low [ ] 0; Med [ ] 2; High [✓] 5
6.5 Possible smolt access out of leat/by-wash (undamaged) — [✓] 0; [ ] -2; [ ] -5
Total 6.4 + 6.5 [5]
6.8 Risk of wild adult access in — [✓] [ ] [ ]

7. EVIDENCE OF ISSUE (Site History)
7.1 Evidence/records of wild juvenile ingress — Y [✓] Proceed with 7.2; N [ ] Collect data

Juveniles
7.2 Evidence/records of parr/smolt ingress — Y [✓] N [ ]
7.3 % of sub catchment parr/smolt run entrained & lost (estimate) — 0-1% [ ] 0; 1-2.5% [ ] 2; >2.5% [✓] 5
7.4 % of whole river catchment parr/smolt run entrained & lost (estimate) — 0-1% [ ] 0; 1-2.5% [ ] 4; >2.5% [✓] 10
Total 7.3 + 7.4 [15]

Adults
7.5 Evidence/records of adult ingress — Y [ ] N [✓]
7.6 % of sub catchment spawning stock entrained & lost (estimate) — 0-1% [✓] 0; 1-2.5% [ ] 2; >2.5% [ ] 5
7.7 % of whole catchment spawning stock entrained & lost (estimate) — 0-1% [✓] 0; 1-2.5% [ ] 4; >2.5% [ ] 10
Total 7.6 + 7.7 [0]

Figure 4.1 Environment Agency checklist for a hypothetical SFFAs.14 site (Turnpenny *et al*, 1998).

4.2.5 Under the Habitats Regulations

Projects within or connected with a designated SAC and therefore regulated under the Habitats Regulations demand more stringent risk assessment criteria. The critical test is whether the *appropriate assessment* can demonstrate no adverse effect on integrity of the European site. In this context, the BATNEEC criteria are replaced by BAT (Best Available Technology), without consideration to cost. The project may not proceed if adverse effect on site integrity cannot be avoided. The appropriate assessment therefore is the form of risk assessment procedure used in such cases.

The Agency's policy is summed in the following extracts from the website (www.environment-agency.gov.uk); note that the powers can apply to new and existing plans or projects that may be impacting upon SACs:

"The Agency will ensure that all applications for new permissions will be screened for potential impacts on European Sites. This applies to all proposals that require Agency approval and is a consolidation of existing statutory obligations to protect SSSIs under the Wildlife and Countryside Act (1981), as amended by the Countryside and Rights of Way Act (2000).

Where a significant effect on a European site is likely, an appropriate assessment will be carried out by the Agency in consultation with EN[22] and CCW[23].

In all cases the determination will be made by the Agency on best available information and taking full account of advice from EN/CCW as agreed in this joint guidance.

The Agency has obligations to review existing consents, licences, permissions and activities that are likely to be having a significant effect on a European Site. This assessment of likely significant effect is in relation to the permission alone or in combination with other permissions or plans or projects. Where a likely significant effect is established, the Agency will carry out an appropriate assessment to determine whether to affirm, modify or revoke the permission.

Under the Review of Consents process, an appropriate assessment will be undertaken for those European sites protected in the UK by means of Regulation 10 (1) of the Conservation (Natural Habitats, & c.) Regulations 1994 as amended by the Conservation (Natural Habitats, & c) (Amendment) (England) Regulations 2000.

The process will affect only those permissions that have been identified as 'relevant' using guidance agreed with EN and CCW, and by those with the legal powers to do so".

The process is well illustrated by the case of Fawley Power Station in Hampshire, an oil-fired plant that has been in operation since the late 1960s. Following the designation of candidate SACs on the Solent (marine habitat) and R. Itchen (salmon river), lying to the south and north respectively of the plant, a new abstraction licence was issued in the year 2002. As a result of an assessment under the Habitat Regulations, to remove the risk of an adverse effect on site integrity, various conditions were attached to the licence, including:

- the installation and operation of an acoustic fish deterrent system at the cooling water intake;

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[22] English Nature

[23] Countryside Council for Wales

- implementation of a fish rescue and return facility at the cooling water drum screens;
- operation of a weekly/monthly fish catch monitoring programme on the cooling water drum screens to estimate total annual fish catch for an indefinite period;
- quarterly sampling of fish by several capture methods at designated reference sites in Southampton water and the Solent, against which to judge any change in fish catch at the plant;
- annual reporting of the findings to the Agency.

5 DESIGNING FOR PERFORMANCE

Most of the information given here is drawn from the various screening guidance documents listed in section 1.

5.1 Timing of Fish Movements

Effective screening must first of all be targeted to the species and lifestages of fish that are to be protected. This will determine the method best suited, the critical times of the year and the specific design details for the fish screen (mesh size, etc.).

Table 5.1 Downstream migrations of some fish species found in UK waters and times of the years when vulnerable to entrainment (Lucas *et al*, 1998)

Fish Species	Migratory Habit	Vulnerable Life Stage	Time of Year
Atlantic salmon (*Salmo salar*)	Anadromous	parr* (8-10cm) smolt (12-15cm) kelt (>60cm)	autumn* spring, autumn* winter
Sea trout (*Salmo trutta*)	Anadromous	parr* (8-10cm) smolt (15-22cm) kelt (>40cm)	autumn* spring, autumn* winter
Twaite shad (*Alosa fallax*), allis shad (*A.. alosa*)	**Anadromous**	descending fry & juveniles spent adults	late summer/early autumn early summer
Smelt (*Osmerus eperlanus*)	Anadromous	fry	early summer
River lamprey (*Lampetra fluviatilis*)	Anadromous	descending juveniles (9-13cm)	winter/spring
Sea lamprey (*Petromyzon marinus*)	Anadromous	descending juveniles (9-13cm)	winter/spring
Brook lamprey (*Lampetra planeri*)	Potomadromous	descending juveniles	autumn
Eel (*Anguilla anguilla*)	Catadromous	descending adults	all year but mainly autumn
Cyprinids	Potomadromous	pinhead fry 0 & 1 gp	spring (post-hatch) summer/early autumn

Seasonal events may allow more focus in the design. It is common practice with smolt screening, for example, to install the screens only during the spring period of the smolt run; at other times of the year they are replaced by coarser screens or bar racks to ease operational problems. It may also restrict the range of water temperatures that need be considered when looking at fish swimming speeds. Most important of all, knowledge of the timing of fish runs will allow options of modulating or temporarily ceasing abstraction to be considered. In some operations, it is found to be more cost-effective to cease or reduce abstraction during critical periods than to install screens.

Temporal modulation may be seasonal (e.g. during the period of a smolt run), daily (e.g. shut-down on days when fish movements have been reported) or diurnal (e.g. cessation at night to avoid nocturnal migrations) (Solomon, 1992). Table 5.1 provides a summary of the seasonality and vulnerable life–stages of UK migratory species.

5.2 Intake Velocities and Fish Swimming Performance

Swimming performance is strongly influenced by the species and the length of the fish and to a lesser extent by water temperature. The required criterion is that the fish approaching an intake should be able to swim fast enough and for long enough to ensure their escape via the bywash or any other route provided to return them to the main river flow. Whether this is achieved by using sustained (aerobic) or burst (anaerobic) swimming will depend on conditions: burst swimming will usually require high motivation by the fish, e.g. a startle response that might be caused by a strong stimulus (e.g. electric shock, sound pulse or strobe light flash).

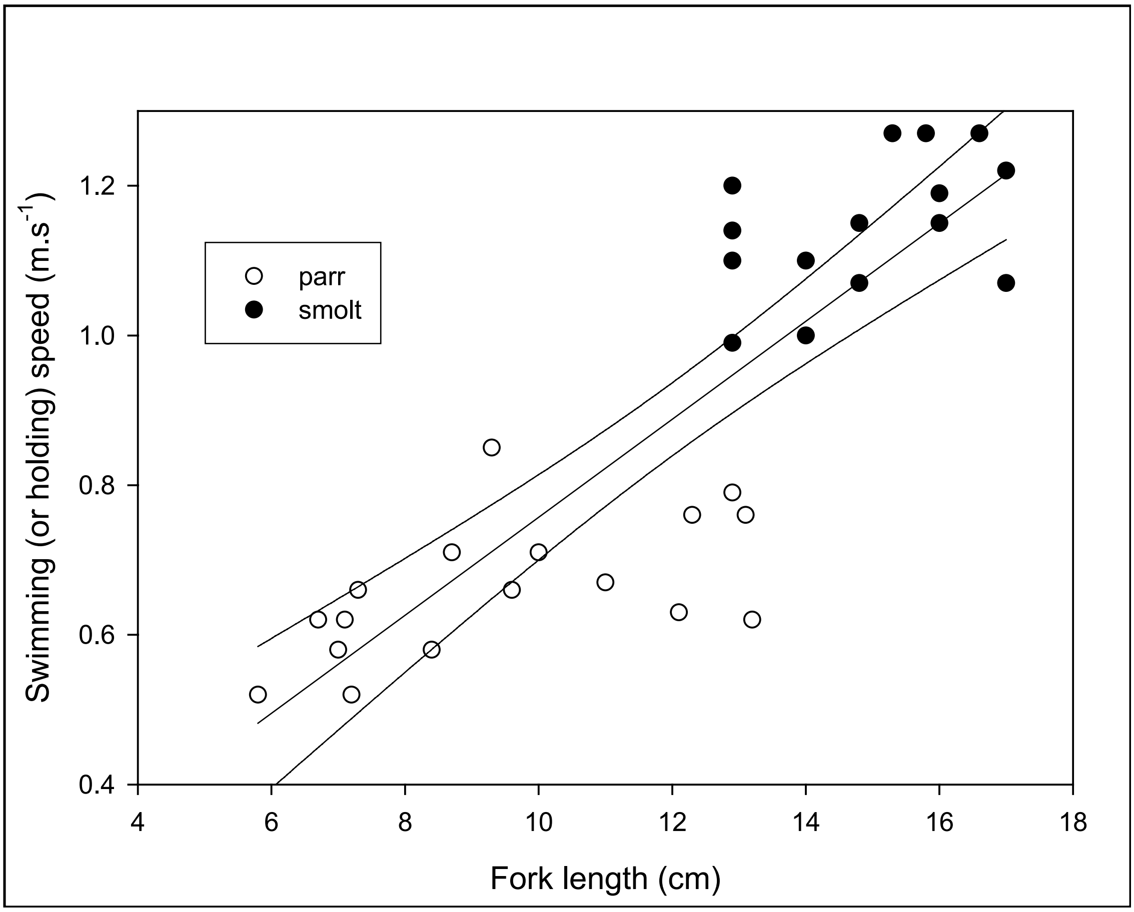

Figure 5.1 The relationship between fork length and swimming speed maintainable for at least 200 min for Atlantic salmon parr and smolts. Dotted lines are 95% confidence intervals (after Peake & McKinley, 1998).

5.2.1 Salmonid Smolts

For salmonid smolts, a swimming speed equivalent to 2 body-lengths per second (bl s^{-1}) has for a long time been taken as the 'design' value for water intakes (Solomon, 1992). While the 2 $bl.s^{-1}$ criterion has been widely used, it is based at least in part upon the notion that the process of smoltification causes some physiological impairment of swimming ability. Solomon (1992) refers to data from Thorpe & Morgan (1978) which indicated that a 12 cm parr could maintain station in current speeds of up to 7 $bl.s^{-1}$, whereas hatchery-reared smolts were unable to maintain station at >2 $bl.s^{-1}$. In fact, their paper refers to position-holding in non-swimming smolt, not smolts in the water column. More recent studies with actively migrating wild Atlantic salmon smolts showed that smolts performed at least as well as parr (Peake & McKinley, 1998; see Figure 5.1) and better than some other salmonids (Peake *et al.*, 1997). Data from Peake & McKinley (1998) indicate that e.g. an average 12 cm smolt could sustain a swimming speed of 7.1 $bl.s^{-1}$ (85 cms^{-1}) indefinitely. Following the Agency's recommendations from the fish swimming speed R & D[24] that the 90th%ile value should be used (i.e. the swimming speed attainable by 90% of the population), when using swimming speeds in fisheries engineering designs, it would be safer to work with a lower-than-average value, e.g. 75 cms^{-1} (equating to the average for a 10 cm parr or smolt); this value is equivalent to the 90%ile value for parr or smolts in the size range 10-15cm. The Peake & McKinley tests were carried out at temperatures of between 12-20^{o}C; while water temperature potentially may influence fish swimming performance, the effect of temperature across this range was not statistically discernible within their dataset. At very low temperatures (i.e. below 5^{o}C) it might be expected that swimming would be cold-impaired but smolt migrations do not commonly occur at such low temperatures. Swimming trials of brown tout conducted under R & D Project W2-026[25] concluded that performance was generally better at temperatures between 5-12^{o}C than at higher temperatures, and the same may be true for salmon and sea trout, both of which are cold-water species. Therefore the 75 cms^{-1} value can be considered a speed that would be sustainable by smolts of either species at sizes and temperatures commonly found in British waters.

5.2.2 Salmonid Kelt

Measurements by Booth *et al.* (in Turnpenny *et al*., 1998) show maximum sustainable swimming speed of 2 $bl.s^{-1}$ for Atlantic salmon kelt at a water temperature of 7^{o}C.

5.2.3 Other Freshwater Fish Species

R & D Project No. W2-049 "Swimming Speeds in Fish" investigated the swimming speeds of various freshwater fish species, including: brown trout, barbel (*Barbus barbus*), grayling (*Thymallus thymallus*), eel, bream, roach, chub (*Leuciscus cephalus*), dace (*Leuciscus leuciscus*) smelt, and adult shad. Data are available for a range of fish sizes (but not fry, at present) and water temperatures. A computer program "SWIMIT (v2.1)" allows swimming speeds to be calculated for each of these species according to body length and temperature. Table 5.1 presents data extracted from SWIMIT (v2.1) for a range of fish sizes and water temperatures. It is recommended that 90th percentile values for *endurance* (not burst) speed should be used to determine design values for intake escape velocity. It is prudent when doing so to allow for the smallest size group of fish likely to be present and the lowest water temperature band. Note that shad have not

[24] Environment Agency National R & D Project W2-026, "Swimming Speed in Fish"

been included in Table 5.1, as it is principally 0-group fish that would be at risk of entrainment and these have yet to be tested.

For a mixed cyprinid population, for instance, from Table 5.1, a maximum escape velocity of 22 cms^{-1} would protect most species at all times of the year and for sizes down to 5 cm length, although a lower value would be required where juvenile bream were at risk. However, the essence of the approach is to be flexible and make use of the detailed information available. The same principle can be applied to the other species shown.

In the case of down-migrating juvenile lampreys, the nearest relevant data is that given by North American workers (Moursund *et al.,* 2003) on Pacific lamprey (*Lampetra tridentata*). In view of current UK interest in lamprey conservation, it is worth taking a brief look at their findings.

Laboratory investigations were carried out into Pacific lamprey impingement on fixed wedge-wire (3 mm spaces) bar screens. At a velocity of zero, individuals were able to swim freely within the test chamber. When this velocity was increased to 45 cms^{-1}, 70% and 97% of fish were impinged after exposures of 1 minute and 12 hours respectively. The tendency of juvenile lamprey to use their tails for locomotion resulted in many individuals becoming wedged between the bar spaces. At velocities <45 cms^{-1}, juvenile lampreys were able to free themselves from the screen surface (Moursund *et al.*, 2003). Figure 5.2 illustrates critical velocities in relation to a 3 mm wedge-wire screen.

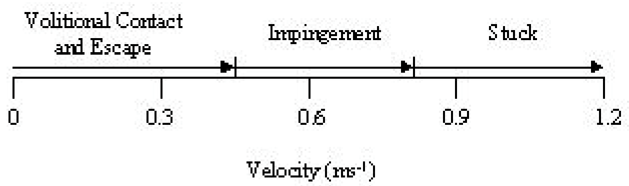

Figure 5.2 Pacific lamprey impingement velocities for a 3 mm wedge-wire screen (after Moursund *et al.*, 2003).

Moursand *et al.* (2003) reported swimming speed tests on 13-cm-long Pacific lamprey juveniles, which gave burst speeds of 70 cms^{-1} (5.2 body lengths per second: $bl.s^{-1}$) and prolonged swimming speeds of 23 cms^{-1} over 5 min and 15 cms^{-1} over 15 min. Water temperature was not stated. They also conducted tests to investigate the performance of different screen materials. After starting with a 3 mm wedge-wire bar screen, they tested one with a 1.75 mm spacing, which gave much improved results, even at velocities of up to 1.2 ms^{-1} (Figure 5.3), since the lampreys were less susceptible to becoming trapped between the bars. Also, they found that lampreys were far more likely to become wedged between horizontally aligned bars than vertical ones. However, the 3 mm bar spacing appeared to be suitable when used with a velocity <45 cms^{-1}.

Species	Test Type	<10 ºC band						10-15 ºC band						>15 ºC band					
		5cm		10cm		20cm		5cm		10cm		20cm		5cm		10cm		20cm	
		Mean	90th	Mean	90th	Mean	90th	Mean	90th	Mean	90th	Mean	90th	Mean	90th	Mean	90th	Mean	90th
Chub	**Sustained**	51	33	81	49	78	29	53	37	85	58	86	48	55	42	89	67	95	66
	Burst	92	50	112	67	133	84	104	58	125	75	145	92	112	64	133	81	153	98
Roach	**Sustained**	25	22	48	37	92	42	25	24	48	40	93	48	25	25	49	44	94	55
	Burst	70	13	103	41	137	70	79	19	113	47	147	76	86	23	120	51	153	79
Bream	**Sustained**	11	9	21	19	42	37	14	12	28	25	55	49	17	15	34	31	68	61
	Burst	121	90	138	107	155	124	100	69	117	86	135	104	87	56	104	73	121	90
Dace	**Sustained**	38	31	45	46	64	29	49	33	52	50	71	36	61	35	58	53	77	43
	Burst	83	44	114	71	144	98	91	49	121	76	151	103	96	53	126	79	156	106
Barbel	**Sustained**	43	26	49	32	60	43	53	32	59	38	70	50	63	39	69	45	80	56
	Burst	141	79	182	120	223	161	161	99	202	140	244	182	175	113	216	154	257	195
Grayling	**Sustained**	28	17	34	21	47	28	35	18	41	21	53	29	41	18	48	22	60	30
	Burst	103	48	136	81	170	115	99	44	132	77	166	111	96	41	130	75	163	108
Brown trout	**Sustained**	45	37	86	66	150	100	42	31	79	55	137	78	39	26	73	44	124	56
	Burst	63	0	11	0	158	30	57	0	104	0	152	21	53	0	100	0	148	15
Eel	**Sustained**			<0.33	<5	10	<5			<0.33	<5	14	<5			8	<5	18	7
	Burst			97	76	111	90			101	80	115	94			103	82	117	96
Smelt	**Sustained**	30	25	36	29	47	36	38	35	44	38	55	45	45	44	51	48	62	55
	Burst	87	80	119	102	18	0	85	78	116	97	11	0	84	75	112	93	4	0

Table 5.1 Burst and sustained speeds of freshwater fish in relation to fish size and temperature, with mean and 90th percentile values (from SWIMIT v2.1)

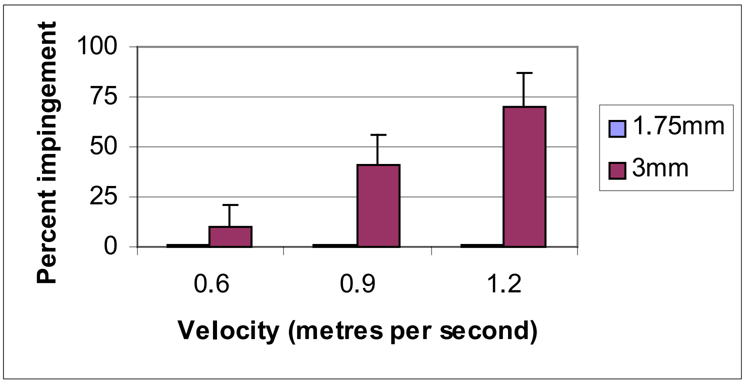

Figure 5.3 Impingement of Pacific lamprey juveniles on a wedge-wire screens with different spacings over a range of approach velocities (after Moursund *et al.*, 2003).

5.2.4 Marine & Estuarine Fish

Turnpenny (1988) provided data to estimate the sustained swimming speeds of a variety of marine and estuarine fish species (Table 5.2) The coefficients shown in Table 5.1 are used in a standard model of the form:

Sustained swimming speed = $(a + b.\mathrm{T})\mathrm{L}^{0.6}$ ... (2),

Where: T is the water temperature, *a* and *b* are species-specific temperature coefficients.

Table 5. 2 Swimming speed coefficients for coastal species (Turnpenny, 1988a).

Species	Coefficients		
	a	b	n
Sprat Herring }	9.3	0.58	285
Cod Whiting Pout Poor Cod }	3.8	0.56	170
Plaice Flounder Dab Sole }	3.8	0.56	170
Bass	6.2	0.82	56
Grey Mullets	6.2	0.82	67
Sand Smelt	5.0	0.55	166
Salmon	8.0	0.32	35

n = number of experimental observations.

These can be used when considering coastal waters, but allowance should be made for the fact that these are mean and not 90th%ile values. It is suggested that the calculated values should be multiplied by 0.66 to obtain a design value approximating to the

90^{th}%ile: this is typically the difference found between mean and 90^{th} percentile values in R & D Project No. W2-049 (Table 5.1)

5.2.5 Channel Velocities and Approach/Escape Velocities

There is some confusion over the terminology used by different authors and the following definitions are proposed (see Figure 5.5):

1. the *channel velocity* is the velocity in front of the screen measured axial to the flow channel;
2. the *escape velocity* is the velocity in front of the screen measured perpendicular to the screen face, irrespective of the screen angle to flow - i.e. it is the minimum velocity at which a fish would need to swim in order to escape. This is also known by some authors as the 'approach velocity'; by convention, this is measured a short distance (e.g. 10 cm) in front of the screen, where a fish might swim, rather than for example between the bars of the screen (Nordland, 1996; Turnpenny *et al.*, 1998).

The screen should be designed to give a maximum escape velocity that is within the swimming performance capabilities of the fish that are to be excluded by the screen (see above). In selecting an appropriate value, allowance should be made for:

- partial clogging of the screen (depending on the level of debris present and the expected efficiency of any screen cleaning mechanism);
- any silt build-up in front of the screen that is likely to occur between routine de-silting operations.

It is recommended that screens should be designed with at least 20% over-capacity to allow for partial blockage or blinding.

5.2.6 Advisory Escape Velocities for Fish Screens

No statutory limits on escape velocities exist at present within the UK and the onus on the operator is to provide a system that avoids injury to fish (Turnpenny *et al*., 1998). The Salmon Advisory Committee (1995) recommended an escape velocity of 25 cms^{-1}, while The Scottish Office (Anon., 1995) suggest 2 body-lengths per second (bl s^{-1}), which might equate to 25 cms^{-1} for a typical Scottish smolt size of 12.5 cm, or 30 cms^{-1} for a 15 cm smolt that might be found further south. Solomon (1992), suggested an escape velocity of 30 cms^{-1} would be appropriate for smolt, while a lower velocity of 15 cms^{-1} would be required where coarse fish were present.

In the light of the published scientific data presented in section 5.2, these values appear too restrictive for many situations and would potentially impose unnecessary construction and maintenance costs on developers and operators of intakes. Where good experimental data exist to develop more informed design criteria, their use is recommended. For most cases it will be appropriate to use the 90^{th} percentile maximum sustainable swimming speed (MSSS), based on the smallest size of fish and the lowest water temperature that the species is likely to encounter at the facility. This value implies that 90% of the fish approaching the screening system would be capable of maintaining that speed for up to 200 min. In a well-designed screening system, which offers adequate guidance cues to steer fish towards the bywash entrance, this period should be more than adequate for fish to pass the screen.

For most purposes, the swimming speed R & D (W2-026) (see Table 5.1) indicates that the following criteria would be suitable (Table 5.3):

Table 5.3 Maximum recommended escape velocities for intake where salmonids or cyprinids are present

Fish Present	Maximum Escape Velocity Perpendicular to the Screen
Salmonids ≥10cm in length, including kelt	75 cms^{-1}
Cyprinids (except bream) ≥ 5cm in length	22 cms^{-1}

Aitken *et al.* (1966) refer to design limits of 0.75-0.90 ms^{-1} for kelt. Based on the swimming speed information given above, which indicates a sustainable speed of 2 bls^{-1} for kelt, the 75 cms^{-1} value for smolts would be a suitable design limit for all Atlantic salmon applications, although smaller sea trout kelt would require a proportionately lower limit.

For fish of different sizes or other species, these values would need to be revised according to available experimental data.

Experimental data are not yet available for juvenile shad, nor for European lamprey species. Where 3 mm wedge-wire screening material is used, the 45 cms^{-1} 'volitional escape' value given by Moursand *et al*. (2002) for Pacific lampreys may be appropriate, but it is recommended that a more conservative design value of 30 cms^{-1} should be adopted until robust experimental data for European species become available. For juvenile shad, sprat/herring data from Table 5.2 should provide a reasonable approximation.

5.2.7 Uniformity of Flow Conditions

In calculating escape velocities, the most common approach used by engineers is to take the average velocity (*U*), as given by the flow (*Q*) divided by the total screen area (*A*):

$$U = \left[\frac{Q}{A}\right] \qquad (1).$$

This provides an initial indication but does not take account of bed and wall friction, which will tend to reduce velocities near to surfaces, leading to higher values in the mid- and upper-channel, nor does it allow for any flow asymmetry caused by inertial effects on bends or angles. Hydraulic modelling can provide useful insights into these aspects and allow an intake design to be developed or modified to achieve the desired flows at the screen surface.

Should the velocity at the screen face always be uniform? Generally it should. The chief purpose in hydraulic design is to avoid high velocity 'hot-spots' that might cause fish to be impinged onto the screen, resulting in possible death or injury (see below).

Attention is also drawn to asymmetrical changes to the flow distribution that can be caused by sedimentation. Ideally, this possibility would be anticipated through the application of hydraulic studies to predict the sedimentation regime.

5.3 Fish Behaviour in Front of Screens

A screen or trash rack represents an obstacle in the flow-path and fish approaching the screen generally turn to face upstream upon meeting it. However, their behaviour in front of the screen depends on the velocity conditions. This effect has been described by various authors (Rainey, 1985; Pavlov, 1989; Turnpenny *et al.*, 1998).

- At **low escape velocities** relative to their swimming capacity (e.g. @<2 bls^{-1}), a school of fish approaching a screen are often seen to circle around upstream of it, searching for an escape route. The leading fish may then locate the bywash entrance and enter it, head first, some or all of the other fish in the school then following it. If the hydraulic or other conditions in the bywash are unsuitable, and the channel velocity is low, the fish may elect to swim back upstream to find another route. The author (AWHT) has observed smolts migrating back up a lade over a distance of 500 m to re-enter the main river when the bywash was unsuitable.
- At **intermediate escape velocities**, close to their maximum sustainable speed, fish will face upstream into the flow and pass tail-first into the bywash (see Figure 5.4, lower diagram).
- At **high escape velocities**, above their maximum sustainable speed, fish will orientate perpendicular to the screen face and pass tail-first into the bywash. This orientation allows the fish to avoid impingement on the screen using the lowest possible swimming speed and is therefore the limiting case for screen design.

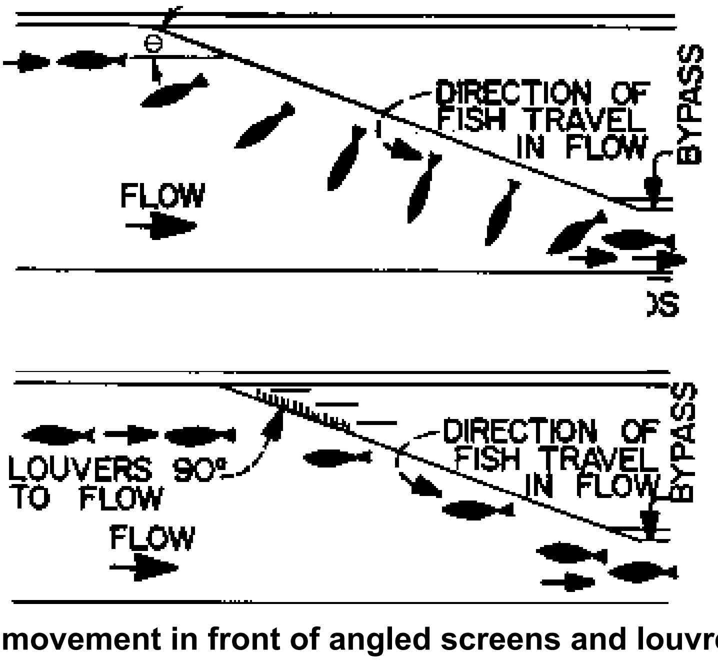

Figure 5.4 Fish movement in front of angled screens and louvres. Lower diagram applies when the channel velocity is below or near the maximum sustainable swimming speed of the fish, upper diagram when channel velocity exceeds the swimming speed of the fish (Rainey, 1985).

5.4 Effect of Screen Angle to Flow

By placing the screen or trash rack at a diagonal angle to the flow (as seen in plan view), fish can be guided to the lower end of the diagonal where a bywash is provided to permit their safe transit downstream. Figure 5.4 again illustrates this principle. Furthermore, the angle of the screen can be set to ensure that the escape velocity is kept below required design value. A screen or trash rack set at a diagonal angle to the flow to bias the fish towards the bywash is always better than one set at right-angles to the flow; in a perpendicular arrangement, no guidance is offered to the fish and this extends the time taken for them to locate the bywash. With large screen arrays, they may become exhausted by the water flow before they can escape.

Figure 5.5 shows the relevant velocity components for an angled fish barrier. The main channel velocity is denoted $\boldsymbol{U_a}$. The velocity perpendicular to the screen face is the fish's escape velocity, $\boldsymbol{U_e}$. For a barrier angle ϕ, this is calculated as:

$$\boldsymbol{U_e} = \boldsymbol{U_a} \sin \phi \qquad (2).$$

The sweeping velocity, ***Us***, is the component parallel to the screen face. This is used to calculate the time taken for the fish to traverse the screen from any given point, when swimming at velocity $\boldsymbol{U_e}$. It is calculated as:

$$\boldsymbol{U_s} = \boldsymbol{U_a} \cos \phi \qquad (3).$$

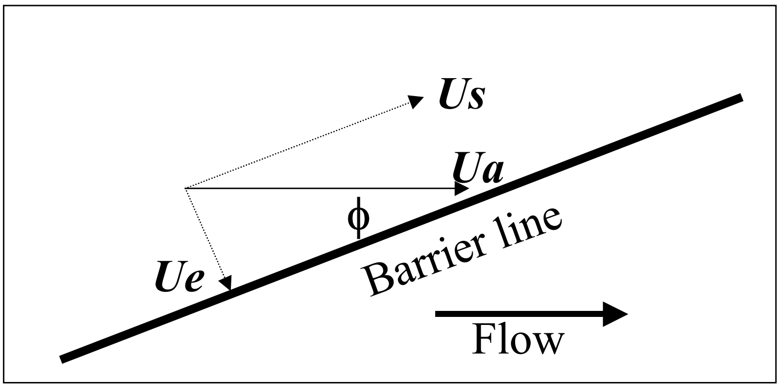

Figure 5.5 Flow velocity components in front of an angled fish barrier. U_a is the channel velocity, *Ue* is the fish escape velocity and *Us* is the sweeping velocity component along the face of the screen (after Turnpenny *et al*., 1998).

For design purposes, the worst case is taken, i.e. the travel time ***T*** (s) for a fish at the extreme upstream end of the screen to reach the bywash. For a screen of length ***L*** m, this is calculated:

$$\boldsymbol{T} = \boldsymbol{L} / \boldsymbol{U_s} \qquad (4).$$

The United States National Marine Fisheries Service recommends that the number of bywashes provided should ensure that the maximum time *T* taken by a fish once it has reached the screen to enter the bywash should be 60s as calculated by this method (Nordland, 1996).

5.5 Selection of Mesh Aperture

The mesh aperture required depends on the size of the fish to be excluded. Turnpenny (1981) gave a formula for computing the rectangular mesh size needed to exclude fish of a given shape and size:

$$M = L/(0.0209L + 0.656 + 1.2F) \qquad (5).$$

Where: M is the square mesh size in mm, L is the fish length in mm (*standard length* - measured from the tip of the snout to the caudal peduncle) and F is the fineness ratio (defined here as the length divided by the maximum depth of the fish). This formula ensures that the calculated aperture size is small enough to exclude a fish by the bony part of the head, i.e. it is not the size at which a fish would just penetrate the mesh.

Values for F in different species are shown in Table 5.1 and Figure 5.6, shows the F values with examples of mesh size *vs*. length of fish excluded. A mesh size commonly used for smolt exclusion is 12.5 mm square; from Figure 3.1 it is seen that this would exclude smolts down to a length of around 12 cm. In Scotland, a rectangular mesh size of 12.5 mm (vertical) x 25 mm (horizontal) is commonplace (Anon, 1995) and is also found widely in England and Wales. Recently, there has been a tendency for regulators to specify smaller mesh (e.g. 10 mm square or 102 x 20 mm rectangular) sizes to protect autumn-run salmonid parr and smaller individuals of other species. As the 12.5 mm (vertical) x 25 mm are found by operators already to be onerous, owing to the hydraulic head loss, the risk of blocking and the frequent cleaning requirement, the requirement for smaller meshes is understandably unpopular and sometimes totally impracticable. This has led to the search for alternative screening methods using self-cleaning technologies or behavioural guidance methods.

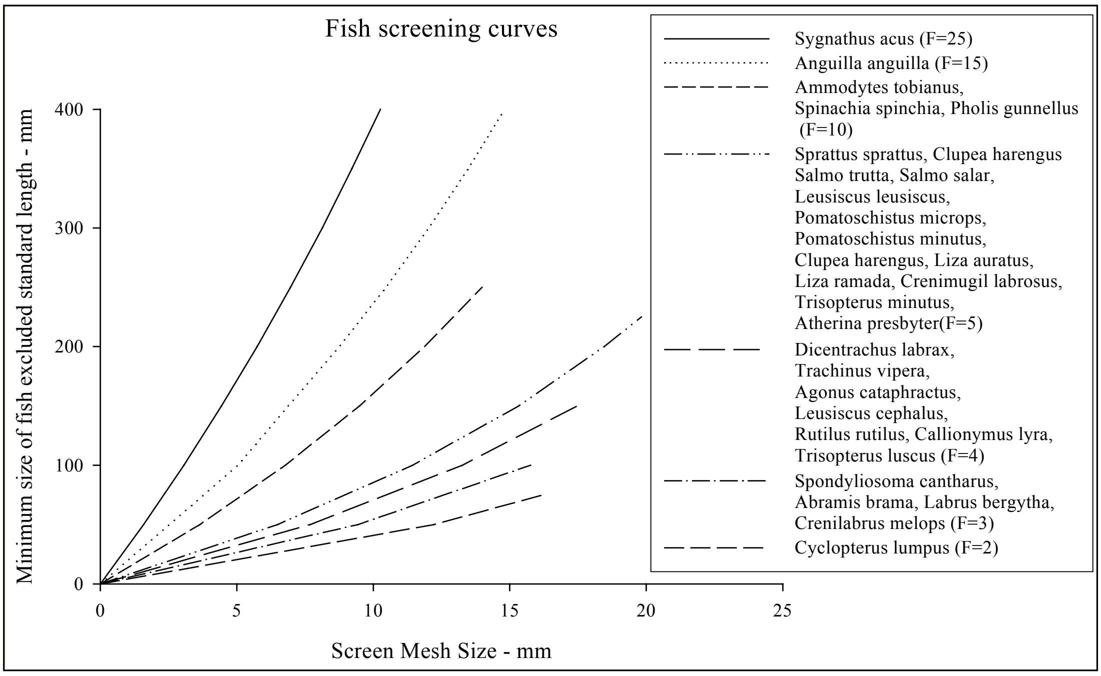

Figure 5.6 Mesh size curves for screening fish of different body shape, plotted from equation 1 (after Turnpenny, 1981, with additional coarse fish data from Table 5.1).

Table 5.1 Observed fineness ratios for 24 marine and freshwater fish species (Turnpenny, 1981, with additional data for cyprinids)

Species	Scientific Name	Fineness Ratio (F)
Bass	*Dicentrachus labrax*	3.67
Bream	*Abramis brama*	2.99
Butterfish	*Pholis gunnellus*	7.73
Chub	*Leusiscus cephalus*	4.39
Dace	*Leusiscus leusiscus*	4.83
Dragonet	*Callionymus lyra*	4.28
Eel	*Anguilla anguilla*	16
Goby (Common)	*Pomatoschistus microps*	5.7
Goby (Sand)	*Pomatoschistus minutus*	5.7
Herring	*Clupea harengus*	4.75
Hooknose	*Agonus cataphractus*	3.87
Lumpsucker	*Cyclopterus lumpus*	2.07
Mullet (Golden)	*Liza auratus*	4.67
Mullet (Thick Lip)	*Crenimugil labrosus*	4.67
Mullet (Thin Lip)	*Liza ramada*	4.67
Pipefish (Greater)	*Sygnathus acus*	25.2
Poor Cod	*Trispoterus minutus*	4.58
Pout	*Trispoterus luscus*	3.92
Roach	*Rutilus rutilus*	3.51
Salmon	*Salmo salar*	4.65
Sand eel	*Ammodytes tobianus*	10.2
Sand smelt	*Atherina presbyter*	5.29
Sea Bream (Black)	*Spondyliosoma cantharus*	2.88
Sprat	*Sprattus sprattus*	4.75
Stickleback (15 spine)	*Spinachia spinachia*	10.9
Trout	*Salmo trutta*	4.37
Weever (Lesser)	*Trachinus vipera*	3.68
Wrasse (Ballan)	*Labrus bergylta*	3.06
Wrasse (Corkwing)	*Crenilabrus melops*	3.06

5.6 Screening for Epibenthic Species

For species that normally live close to the bed, the simple precaution of raising the invert or threshold of any horizontal intake opening may help in reducing entrainment. In a recent survey carried out to determine the optimum opening level for an intake in an estuarine location, different trawling gears were used to compare catch rates just below the water surface and at bed level (A.W.H.Turnpenny, Babtie Aquatic, unpublished data). This was intended to simulate abstraction from a floating pontoon-mounted intake versus a fixed bed-level intake. Predicted fish entrainment levels were reduced by 50% for the floating intake compared with the bed-mounted option. The composition of the catch shifted from one dominated by flatfish and gobies to principally pelagic species.

This concept is also valid for freshwater intakes to reduce the risk to species such as bullheads and loaches. Heuer and Tomljanovich (1979), investigating factors affecting impingement of fish on vertical screens found that provision of a bottom refuge made by blanking off the lower 9 cm of the screen significantly reduced the risk to species living near to the bed.

5.7 Bywash Design Criteria

The placement of a screen flush at the entrance to a channel avoids the need for a bywash, but a sweeping flow is then required to carry fish downstream. Spillway screens (Coanda-effect, Smoltsafe™) require no dedicated bywash structure as such, but do require a surplus flow to convey fish downstream. The bywash must be regarded as integral part of the screening system. A good bywash requires thoughtful design and verification of performance.

5.7.1 Bywash Location

Location of the bywash entrance is important. The entrance to a bywash should be positioned so as to maximise the chances of fish locating it. For an angled screen arrangement, it should be located at the downstream end, in the cleft formed by the screen and the bank or channel wall (see Figure 5.4). The opening should be no more than a metre or two upstream of the screen face. For very large screen arrays, there is a risk that fish may become exhausted or disorientated before fully traversing the screen, in which case, additional bywash entrances would need to be provided at intervals along the screen face. It is unlikely that this would be necessary for screen arrays less than a hundred metres in length, provided that the escape velocity was kept within the sustainable swimming speed limits of the fish, and that there were no structures such as piers getting in the way. At hydropower plants, the bywash entrance should avoid areas of turbulence and plunging water flows near to the turbine inlets, which may make the entrance difficult for fish to detect, and high levels of underwater noise close to the turbines.

5.7.2 Entrance Design

The hydraulic conditions at a bywash entrance are critical to fish diversion efficiency. Rapid changes of velocity and turbulence may cause fish to avoid entering the bywash (Ruggles and Ryan, 1964; Rainey, 1985; Travade and Larinier, 1992). Transitions should be hydraulically efficient, using e.g. a bellmouth entrance design. Haro *et al*. (1998) compared a bellmouth entrance with a simple sharp crested weir design. The water in the bellmouth was accelerated smoothly to a maximum value of 3 ms^{-1} at a rate of 1 ms^{-1} per metre length. Within the first 30 min after release, significantly more Atlantic salmon smolts passed through the bellmouth design than the sharp-crested version. Rapid passage of the bywash is important in reducing the risk of fish entrainment with behavioural barriers or of impingement with mechanical screens. The use of a high entrance velocity reduces the risk of fish turning around and swimming back out. The passage rate of juvenile American shad (*Alosa sapidissima*) was also tested, but found not to be different for the two designs, which would suggest that this species is less influenced by flow conditions. In both species, use of the bellmouth design reduced the tendency of shoals of fish to break up before passing, which suggests that behaviour is less disturbed.

The success of a bywash is strongly influenced by the amount of flow used. The larger the flow, the more likely the fish are to enter it. The US Fish and Wildlife Service call for a minimum bywash attraction flow equating to 2% of the turbine capacity where the screen is oblique to the flow, rising to 5% where the screen is perpendicular to the flow (Odeh and Orvis, 1998). Although there is inevitably a limit on the amount of water that can be allocated to this purpose within any abstraction scheme, it can be economic in hydropower schemes to pump back attraction water after it has fallen by only a small fraction of the nett scheme head (Odeh and Orvis, 1998).

The relative velocities at the bywash entrance and in the main channel are critical. For louvre screen designs, the ratio should be 1.2 -1.4 (Bates and Visonhaler*,* 1957), while Rainey (1985) recommended a value of 1.0 for general use. He proposed that dam-board slots should be provided at the bywash entrance to allow control of the entrance velocity for different flows. When this is done it is preferable to use inserts having an efficient hydraulic lip profile, so that turbulence is kept to a minimum.

Rainey (1985) reported finding bywash arrangements with entrance sizes starting as small as 50 to 150 mm. Openings this small were not attractive to fish and too easily became blocked by debris. His recommendation was to provide a slot to the full depth of the channel, with an entrance width of 300-600 mm. Flow can be regulated by a telescopic weir gate, set back from the entrance, or, in low-cost installations, by dam-boards located in slots across the channel.

Given the sensitivity of bywash effectiveness to good entrance hydraulics, it is recommended that bywash entrances should be 'soft'-engineered from materials such as timber or stone-filled gabions, such that the entrance profile can be readily modified and fine-tuned to give optimum bywash performance. This requires detailed visual monitoring of fish behaviour at the entrance, e.g. by closed-circuit television monitoring.

5.7.3 Light and Visual Attributes

Smolts are reluctant to enter darkened culverts during downstream migration (Rainey, 1985). The same appears to be true of non-salmonid species (percids and clupeids) that have been investigated in Australia (Mallen-Cooper, 1997). Fish tend to resist entry into any form of bywash, such as an orifice or pipe that does not admit light. Open-topped bypass channels are therefore the preferred solution.

The visual appearance of a bypass, as seen from the fish's eye view, is important. Any apparent discontinuity of surroundings may cause fish to turn back, reducing or delaying passage. Fish moving from open water and meeting a visible structure will generally turn around and face upstream, as a result of the optomotor reflex (Harden-Jones, 1967; Arnold, 1974). In experiments, Haro *et al*. (1998) showed that smolts and shad mostly displayed this behaviour when entering a bypass, up to the point when they became exhausted and passed downstream. This was the case even at very low light levels (<0.1 lux), indicating the persistence of visual cues, or detection of the flow field or detection of displacement by another sensory system. Reduction of visible discontinuities from the fish's aspect may therefore improve bypass efficiency, for example blending the colour of the bypass entrance into its surroundings. Flat grey colour is often used for this purpose on any painted surfaces, although biofouling will soon naturalise most surfaces. Inspection using an underwater television camera or diver may be helpful.

Artificial lighting has been used to enhance bypass attractiveness. In an early study, Fields *et al.* (1958) found that juvenile salmonids were repelled by bright light but attracted by dim light. On the other hand, Larinier and Boyer-Bernard (1991) found that passage rates of Atlantic salmon smolts through a bypass at night increased when adjacent lights were turned off, presumably owing to the loss of visual cues. Illumination has been observed to enhance bypass efficiency for juvenile American shad (Anon., 1994). As no clear message emerges from these studies, no specific recommendation can be made. Given the ease and low cost of trying out lights, some experimentation may be worthwhile.

5.7.4 Bywash Conduits and Outfalls

Fish handling within the bywash and at the return point should be as gentle as possible, avoiding sharp bends (3 m minimum radius), sudden drops, and rough surfaces and irregularities that might cause abrasion. This is particularly important for smolts, which have loose scales and become vulnerable to osmotic disorders upon scale loss. The maximum scale loss tolerated by smolts is of the order of 20-30% (Kostecki *et al*., 1987). Open, half-round channels are preferred.

Even steep chutes have proved successful, provided that there is adequate water depth at the receiving end. The smolt return chute at Dunalastair Dam (Scottish HydroElectric plc), which is in the form of an open channel some 15 m long and angled at 45^o to the vertical, functions well, with no evident harm to smolts. It is important that the fish are not dazed or disorientated at the point of return, which would make them more vulnerable to predators. The risk of predation of returned fish is a weak point in any diversion scheme and has received little investigation. Odeh and Orvis (1998) give the following criteria for the plunge pool from a return chute:

- plunge pool volume: 10 m^3 per cumec of bywash flow;
- plunge pool depth: ¼ of the differential head but no less than 0.9 m for head differences of <3.6 m;
- at tailraces, the chute elevation should be 1.8-2.4 m above the free surface level to avoid adult fish jumping into the chute.

6 SELECTING THE BEST SCREENING SOLUTION

6.1 The Selection Process

Given the wide variety of screening applications and environments and the need to consider the protection of a much-enlarged list of fish species than perhaps in the past, the developer or operator is faced with a potentially bewildering array of options. From the foregoing sections, it should be evident that the following questions must be addressed:

- What is the motivation for fish screening (e.g. statutory/planning requirement, desire to improve environmental performance, 'good-neighbour' policy, etc.)?
- What species and lifestages are to be protected and at what times of the year?
- What level of protection is required under BAT/BATNEEC[25] principles (establish via risk assessment/consultation)?
- What screening techniques will achieve the above cost-efficiently and within the environmental and engineering constraints of the site and with due regard to public safety?
- How will the screening system be maintained, taking account of health and safety issues for the operator?
- What provisions should be made to demonstrate that the screens are working effectively and are being operated and maintained in a way that consistently achieves the required level of performance?

The overall process is shown by the flowchart given in Figure 6.1. This sets out the main steps in developing a fish protection solution for a water intake. Consultation with the various bodies shown is the most important aspect of the whole process. Discussing the issues with the relevant parties at an early stage avoids misunderstandings and can save much time, trouble and cost.

Table 6.1 provides a summary of techniques that, from current knowledge, are likely to provide suitable screening solutions for different applications/environments for the various categories of fish of concern. Various techniques may be shown for each case; the options that are most likely to be suitable are shown in emboldened typeface. In selecting a technique, the issues of required performance, engineering and environmental suitability, public and operator safety and cost-benefit listed above must be taken into account. This will be highly site-specific and a matter of skilled professional judgement. The Environment Agency, as regulator, will not be able to advise on the selection of techniques and it is the responsibility of the operator to consult fully and take any necessary professional advice in this matter. Table 6.1 is not comprehensive and there inevitably remain at this time gaps in our knowledge and uncertainties in the performance of certain techniques with particular fish species and applications/environments. There are also circumstances where there is no established reliable solution at the present and where further research and evaluation will be required.

[25] Best Available Technology/ Best Available Technology Not Entailing Excessive Cost

Figure 6.1 Flow Chart for Planning a Fish Screen

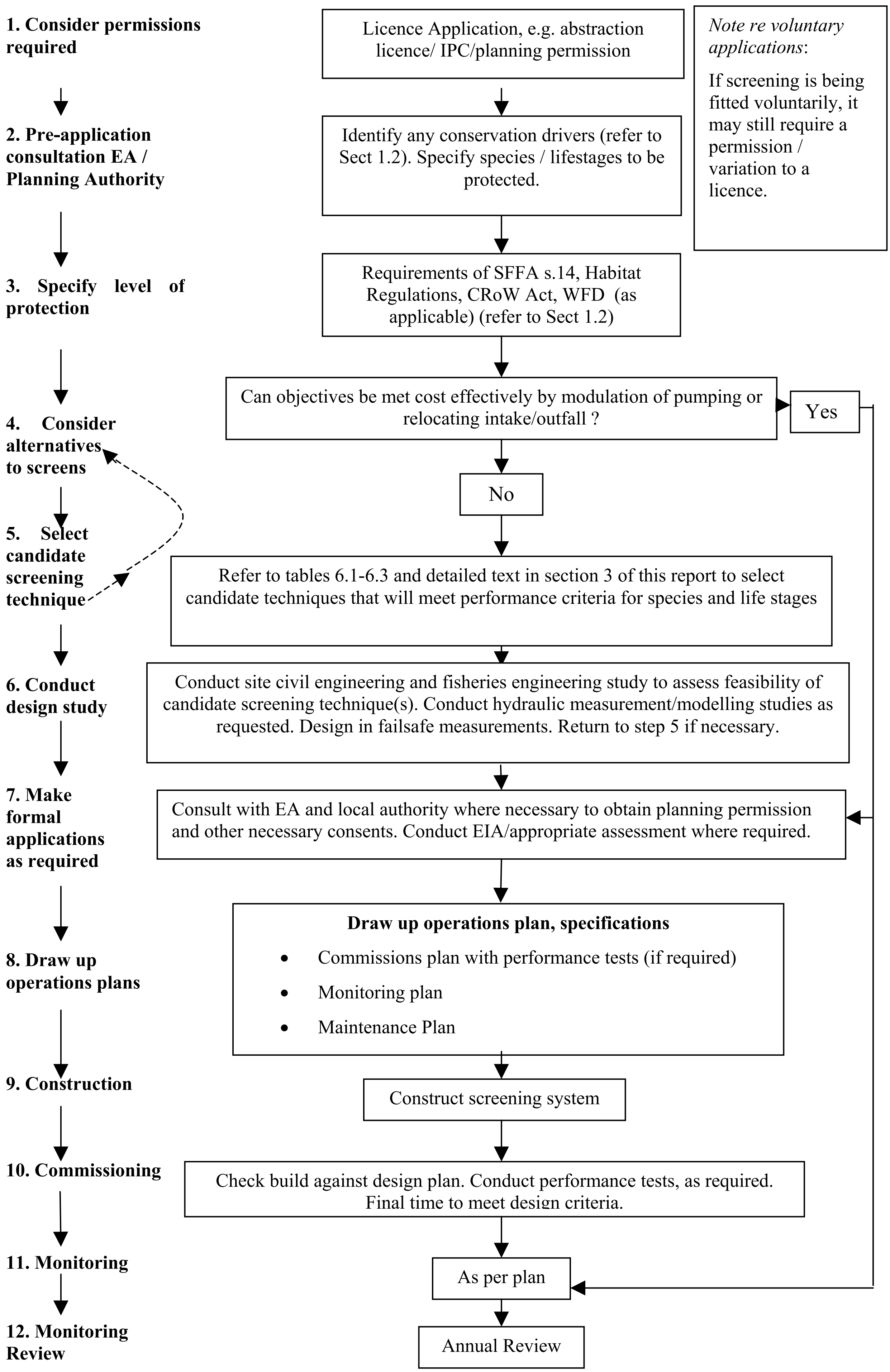

Table 6.1 Screening Techniques Suitable for Freshwater Sites

Freshwater						
Fish species	**Life stage**	**Canal / Industrial / Potable supplies & Fish farms**	**Thermal Power Plant**	**Hydro Electric Power Plant: Low Head**	**Hydro Electric Power Plant: High Head**	**Outfalls**
Salmonids and Coregonids	Juvenile	**Passive Mesh, PWWC, Sub-**gravel intakes, MLES, Acoustic, Light, Bubble	**Passive Mesh, PWWC, Coanda,** Acoustic, Light, Bubble	Acoustic, Light, Bubble	**Passive Mesh, Coanda,** Acoustic, Light, Bubble	
	Smolts	**Passive Mesh, Vertical/inclined Bar rack, Coanda, PWWC,** Sub gravel intakes,Rotary Discreen, Modular Inclined, **Louvre, Acoustic, Light,** Bubble	**Passive Mesh, Vertical/inclined Bar rack, PWWC, Coanda,** Rotary Discreen, Modular Inclined, Louvre, **Acoustic, Light,** Bubble	**Passive Mesh, Vertical/inclined Bar rack,** Louvre, **Acoustic, Light,** Bubble	**Passive Mesh, Vertical/inclined Bar rack, Coanda,** Modular Inclined, Louvre, Acoustic, Light, Bubble	
	Adult	**Passive Mesh, Vertical/inclined Bar rack,** Coanda, PWWC, Rotary Discreen, Sub-gravel intakes, Modular Inclined, Louvre, Electric, Acoustic	**Passive Mesh, Vertical/inclined Bar rack, PWWC, Coanda,** Rotary Discreen, Modular Inclined, Louvre, **Acoustic, Light,**	**Passive Mesh, Vertical/inclined Bar rack,** Louvre, **Acoustic, Light,**	**Passive Mesh, Vertical/inclined Bar rack, Coanda,** Modular Inclined, Louvre, Acoustic, Light,	**Vertical/Inclined Bar Rack, Electric**
Lampreys	Juvenile	**Passive Mesh, Coanda, PWWC,** Sub-gravel intakes	**Passive Mesh, Coanda, PWWC**	Electric	**Passive Mesh, Coanda**	
	Adult	**Passive Mesh, Coanda, PWWC,** Rotary Discreen, Sub-gravel intakes, Modular Inclined, Electric, Deep channel bypass?	**Passive Mesh, Coanda, PWWC,** Rotary Discreen, Modular Inclined, Electric, Deep channel bypass?	**Passive Mesh,** Electric?, Deep channel bypass?	**Passive Mesh, Coanda,** Modular Inclined, Electric, Deep channel bypass?	**Vertical/Inclined Bar Rack, Electric**
Eels/Elver	Elver	**PWWC, Sub-gravel intakes,** MLES, Modular Inclined, Electric, Light	**PWWC, Sub-gravel intakes,** MLES, Modular Inclined, Electric, Light	Electric?, Light	**Coanda,** Electric, Light	
	Adult	**Passive Mesh, Coanda, PWWC**, Rotary Discreen, Sub-gravel intakes, Modular Inclined, Electric, **Light,** Deep channel bypass?	**Passive Mesh, Coanda, PWWC**, Rotary Discreen, Modular Inclined, Electric, **Light,** Deep channel bypass?	**Passive Mesh,** Electric, **Light,** Deep channel bypass?	**Passive Mesh, Coanda,** Modular Inclined, Electric, Light, Deep channel bypass?	**Vertical/Inclined Bar Rack, Electric**
Freshwater coarse fish	Juvenile	**Passive Mesh, PWWC, Coanda,** Sub-gravel intakes, MLES, Acoustic, Light, Bubble	**Passive Mesh, PWWC, Coanda,** Acoustic, Light, Bubble	Acoustic, Light, Bubble	**Coanda,** Acoustic, Light, Bubble	
	Adult	**Passive Mesh, Vertical/inclined Bar,** PWWC, Rotary Discreen, Coanda, Sub-gravel intakes, Modular Inclined, Louvre, **Acoustic,** Light, Bubble	**Passive Mesh, Vertical/inclined Bar,** PWWC, Rotary Discreen, Coanda, Modular Inclined, Louvre, **Acoustic,** Light, Bubble	**Passive Mesh, Vertical/inclined Bar,** Louvre, **Acoustic,** Light, Bubble	**Passive Mesh, Vertical/inclined Bar,** Coanda, Modular Inclined, Louvre, Acoustic, Light, Bubble	**Vertical/Inclined Bar Rack, Electric**
Freshwater benthics	Juvenile	**Passive Mesh, Coanda, Blank off bottom,PWWC**, MLES, Light, Bubble	**Passive Mesh, Coanda, Blank off bottom,**PWWC, Light, Bubble	**Blank off bottom,** Light, Bubble	**Passive Mesh, Coanda,** Light, Bubble	
	Adult	**Passive Mesh, Vertical/inclined Bar,** Rotary Discreen, Coanda, PWWC, **Blank off bottom,** MLES, Modular Inclined, Light, Acoustic	**Passive Mesh, Vertical/inclined Bar,** Rotary Discreen, Coanda, PWWC, **Blank off bottom**, MLES, Modular Inclined, Light, Acoustic	**Passive Mesh, Vertical/inclined Bar, Blank off bottom,** Light, Acoustic	**Passive Mesh, Vertical/inclined Bar, Blank off bottom,** Coanda, Modular Inclined, Light, Acoustic	

Key: Items in emboldened typeface are the most suitable choices but those shown in standard typeface may be suitable in some applications

<table>
<tr><th colspan="6">Marine/Estuarine</th></tr>
<tr><th>Fish species</th><th>Life Stage</th><th>Large Thermal Power Plant with Onshore Intake</th><th>Large Thermal Power Plant with Offshore Intake</th><th>Small Thermal Power Plant / Desalination / Refineries</th><th>Outfalls</th></tr>
<tr><td rowspan="2">Salmonids</td><td>Smolts</td><td rowspan="14">PWWC screens where feasible; Acoustic, Bubble with Fish Return System otherwise.</td><td rowspan="14">PWWC screens where feasible; Velocity Cap with Acoustic, Bubble with Fish Return System otherwise. Keep opening above bed level.</td><td rowspan="14">PWWC alone or Acoustic, Bubble with Fish Return System</td><td></td></tr>
<tr><td>Adult</td><td>30-50 mm-spaced Bar Rack</td></tr>
<tr><td rowspan="2">Shads</td><td>Juvenile</td><td></td></tr>
<tr><td>Adult</td><td>30 mm-spaced Bar Rack</td></tr>
<tr><td rowspan="2">Lampreys</td><td>Juvenile</td><td></td></tr>
<tr><td>Adult</td><td></td></tr>
<tr><td rowspan="2">Eels/Elver</td><td>Elver</td><td></td></tr>
<tr><td>Adult</td><td></td></tr>
<tr><td rowspan="2">Marine/estuarine benthic</td><td>Juvenile</td><td></td></tr>
<tr><td>Adult</td><td></td></tr>
<tr><td rowspan="2">Marine/estuarine demersal</td><td>Juvenile</td><td></td></tr>
<tr><td>Adult</td><td></td></tr>
<tr><td rowspan="2">Marine/estuarine pelagic</td><td>Juvenile</td><td></td></tr>
<tr><td>Adult</td><td></td></tr>
</table>

Table 6.2 Screening Techniques Suitable for Marine Sites

6.2 Multiple Solutions and Non-Screening Solutions

It is stressed again that screening is not always the best solution. It may be more economic and/or protective to modulate abstraction to avoid seasons, days or times of the day when fish are most at risk.

In the case of larger coastal abstractions where dozens of species may have to be protected, a single solution may not be adequate. The best solution found to date for sites where PWWC screening is impractical uses a combination of an acoustic fish deterrent, which is very effective against pelagics and moderately effective against demersal species, and a fish return system which is moderately effective for demersal species and offer the best solution for robust benthic flatfishes and rock fishes. The systems therefore complement each other well. This combination of methods is used at several power plants, including Great Yarmouth, Shoreham, Fawley in England and Doel in Belgium. There are numerous other technology combinations that have shown promise and which should not be overlooked. These include bubble/acoustic/strobe-light combinations, electric and acoustic screens, screen and eel-bypass combinations and use of turbulent attraction flow to improve guidance in front of screens. Reference to Table 6.3 may suggest other techniques that could be combined to achieve the best results for the species present.

6.3 Costs of Different Screening Solutions

The costs of installing fish screens or barriers are highly site-specific and will depend on whether the application is new-build or retrofit, what existing structures are present, what ground conditions are like, the degree of exposure to flood- and other damage, whether power (if required) is available and many other factors. Table 6.4 attempts to provide indicative costs of some of the main techniques in use described in section 3 of this Guide. In most cases the costs are for the screening/ barrier hardware only and exclude costs associated with planning and design, consultancy, site investigations and preparation, installation, commissioning and testing. These likely additional costs may inflate the overall project cost considerably.

	Type of Screen	Relevant section in text	Salmonids	Shads	Eel & Lampreys	Cyprinids & Other Freshwater Fish	Marine		
							Pelagic	Demersal	Benthic
Physical screens	Passive Mesh/Wedge-Wire Panels	3.1.1 & 3.2.2	*****	*****	*****	*****	*****	*****	*****
	Vertical or Inclined Bar Racks	3.1.2	*****	*****	**	***	*****	*****	*****
	Rotary Disc Screen	3.1.3	****	?	?	***	NS	NS	NS
	Smolt Safe'	3.1.4.2	**	NS	?	?	NS	NS	NS
	Coanda Screen	3.1.4.1	*****	NS	*****	*****	NS	NS	NS
	PWWC Screens	3.2.1	*****	*****	*****	*****	*****	*****	*****
	Marine Fish Exclusion System	3.2.4	***	***	***	***	NS	NS	NS
	Modular Inclined Screen	3.3.1	****	NS	****	****	?	?	?
	Labyrinth Screen	3.3.2	*****	****	****	***	?	?	?
Behavioural screens	Louvre Barrier	3.4.2	**	?	NS	?	NS	NS	NS
	Bubble Curtain	3.4.3	**	**	NS	**	**	**	*
	Electric Screen (GFFB)-intakes	3.4.4	?	?	?	?	NS	NS	NS
	Electric Screen (GFFB)-outfalls	3.6.3	****	?	****	****	NS	NS	NS
	Acoustic (SPA/Infrasound)	3.4.5	****	****	**	****	*****	****	*
	Acoustic (BAFF)	3.4.5.8	****	?	NS	****	NS	NS	NS
	Acoustic (Ultrasound)	3.4.5.11	NS	****	NS	NS	***	NS	?
	Continuous Light	3.4.6.1	**	**	**	**	NS	NS	NS
	Strobe Light	3.4.6.2	**	***	***	***	NS	NS	NS
	Eel Deep Bypass	3.5.3	NS	NS	***	NS	NS	NS	NS
	Surface collectors	3.5.2	***	?	?	***	NS	NS	NS
	Velocity Cap	3.4.7	**	***	NS	NS	***	**	?
	Turbulent Attraction Flow	3.5.1	**	?	?	?	?	?	?

NS

Key: note that the ratings assume that the systems are designed using the appropriate criteria for the application. Not Suitable ** Low efficiency *** Suitable for some lifestages ****Suitable for most lifestages *****Excellent for most or all lifestages

Table 6.3 Suitability of Screening Techniques for Different Types of Fish

Table 6.4 Approximate purchase costs (£k) for fish screens and barriers. Costs are for equipment only and exclude installation except where otherwise indicated.

Screen or Barrier Type	Inland		Estuarine/Marine		
	≤1 m^3s^{-1}	10 m^3s^{-1}	≤1 m^3s^{-1}	10 m^3s^{-1}	50 m^3s^{-1}
Positive Exclusion Screens					
Flat Mesh Panel, 12mm	24	50	30	-	-
PWWC Screen, 3mm	50	285	70	430	-
MLES™	160*	1600*	-	-	-
Under-Gravel Filter	160	-	-	-	-
Raked Bar Screen	40	250	40	250	-
Coanda-Effect	13-17	-	-	-	-
Smolt-Safe™ Screen	15		-	-	-
Rotary Disk Screen	130	-	-	-	-
Behavioural Screens					
Bubble Curtain	5	15	5	15	75
Louvre Screen	24	50	-	-	-
Continuous Light	5	20	-	-	-
Strobe Light	10	40	-	-	-
SPA Acoustic Barrier	15	40	15	40-60	80-250
BAFF™ Acoustic Barrier	18	40	-	-	-
Electric GFFB	10	18	-	-	-

* MLES figures are manufacturers estimated fully installed costs in pounds sterling.

Figures given are based on prices obtained from manufacturers. Three scheme sizes (1, 10 and 50m^3) have been considered where appropriate – generally the smaller schemes are relatively more costly owing to fixed minimum costs. Also, inland and coastal sites have been differentiated. This is important because different materials may be required for use in saline waters (e.g. marine grade stainless steel or Cu/Ni alloy), which increases costs. No figure is shown in cases where screens are considered unsuitable.

7 MONITORING OF SCREEN EFFECTIVENESS: RECOMMENDATIONS FOR FURTHER WORK

7.1 Introduction

From the review of screening technologies presented in section 3 of this Guide, it is clear that many different approaches exist and that there has been much innovation in recent years. The development of new techniques reflects the need to provide cost-effective solutions to suit an ever-widening range of circumstances. Often, a technique has been developed for a specific application but if the results look promising others will want to try it in a different situation. Indeed, it may be arguable that every situation is different and that the performance of every new fish screening system installed should be evaluated at the commissioning stage. In practice, comprehensive scientific testing can be very costly and it makes sense to first answer basic questions on effectiveness from soundly designed generic studies. The site-specific questions hopefully can then be addressed in a much more concise test programme where the regulator deems this necessary. The nature of work appropriate to generic trials and to site-specific commissioning trials is discussed below.

Solomon (1992) proposed that trialling of different fish screens would be best carried out at a purpose-built facility, e.g. on a disused mill leat. He pointed out a number of merits of this approach, e.g. that it would not interfere with operation of an existing abstraction, there would not be issues of operators being unwilling to cooperate with a project aimed at identifying the harm that they are doing and that there would not be operational constraints on the manipulation of flow conditions. While these remain valid issues, the breadth of environments and fish species now being considered mean that a variety of such test sites would be required, covering upland and lowland rivers, canals, lakes/reservoirs and estuarine/coastal locations. Also, it is the experience of the present authors that many site owners recognise the benefits to future business of work being undertaken at their facilities and are willing to cooperate. Cooperation often extends not only to providing access to sites but also to assisting with test facilities and being prepared to manipulate flow conditions to suit experiments. In some cases, successful approaches have been made through trade associations, a route which is to be commended where possible. In particular, operators have shown most willingness to cooperate with regard to trialling behavioural screens, since where they prove suitable, their capital and operating costs would often be lower than for positive exclusion fish screens. Further opportunities may arise at newly developed operational sites when trialling is required as a condition of the planning consent or abstraction, impoundment or land drainage consent.

7.2 Priorities for Generic Trials

The number of techniques now in use could create an almost unlimited agenda for testing in order to cover the different environments, species and lifestages and the possible combinations of techniques. Rather than produce an exhaustive list of possibilities, the aim here is to identify where resources would be best directed to meet current needs. Table 7.1 lists the most promising techniques and identifies priorities for trials along with possible test locations/environments. The choice of techniques is necessarily subjective.

Table 7.1 Proposed Generic Trials Required for Different Screening/Guidance Techniques

Technique	Trial Requirements	Suggested Environment or Location
Flat panel screens	None - well proven	n/a
Vertical bar screens	None - well proven	n/a
Wedge-wire panels	Automated cleaning mechanisms for large installations	River or estuarine site
PWWC screens	Ability to self-clean in different environments	Thames Tideway, Beckton (in progress 2004-5) Stillwaters, especially canals
Coanda-effect screens	None - well proven	n/a
Rotary disc screens	Not currently a high priority	n/a
MLES™ barrier	Await outcome of further trials in USA	n/a
Louvre screens	Suitability for non-salmonids	Small hydropower intake
Strobe lights/bubbles	High priority for eels	Small hydropower intake
Acoustic SPA Acoustic infrasound	Well proven for estuarine power plant; more trials on freshwater needed	Potable water intake
Acoustic BAFF™	Trialled at several locations in UK, Europe and USA but more exhaustive testing needed for a range of fish and for depths >3m	Small hydropower intake
GFFB electric	Merits testing on intakes especially for eel and adult lamprey exclusion	Small hydropower intake
Turbulent attraction flow	Merits testing in conjunction with other physical and behavioural methods to improve guidance efficiency	Small hydropower intake
Adult eel bypass	High priority for eels	Small hydropower intake

7.3 Scope of Work and Costs for Generic Trials

7.3.1 Measures of Performance

Fish screens usually form only part of an overall fish diversion or protection system and it is the performance of the entire system that needs to be proven. Turnpenny *et al*. (1998) used the concept the *scheme passage rate* (*SPR*), defined as:

$$SPR\ \% = 100 * N_{leaving} / N_{approaching,}$$

Where $N_{approaching,}$ and $N_{leaving}$ are the numbers of fish approaching and passing the scheme respectively. Although originally referring to hydroelectric schemes, the definition will apply to any kind of intake or outfall scheme. This concept assumes that all the fish are uninjured when passing the scheme. Where there is a possibility of fish being harmed as a result of the scheme, this needs to be taken into account. Examples of where fish might be harmed are:

- during passage through a pump or turbine;
- as a result of contact with screens or other mechanical components of the system;
- through predation associated with behavioural changes that make fish an easier target for predators.

This can be represented by the following expression:

$$SPR'\ \% = 100(1-P_i) * N_{leaving} / N_{approaching,}$$

Where P_i represents the probability of fish death or injury. Note that within this expression, *SPR'* takes the same value whether a fish is injured or killed – i.e. the worst case is assumed, that any fish injured will not survive.

While the *SPR'* is the most important measure of a screening system's performance, another important aspect is the time taken for fish to pass the scheme. This applies principally to off-channel schemes, e.g. hydroelectric schemes where water is diverted through a mill-leat before being returned to the main channel. Delays can lead to increased risk of predation, but this risk is accounted for in the *SPR'* expression. There may be less clear consequences of delays, for example delayed passage downstream might put them at some ecological disadvantage. It is not realistic to propose targets for passage time, as much will be site-specific and will also depend on the physiological state of the fish, water conditions, time of day, etc. Such effects are virtually impossible to define and the pragmatic course of action is to aim to minimise the time of passage of a scheme. As part of any off-channel screen trial it is therefore recommended that the time taken for fish passage should at least be recorded so that comparisons may be made for different operating conditions or with alternative screen systems tested under comparable conditions.

7.3.1.1 Test Methods

Methodologies for trials should be aimed at estimating the scheme passage rate (*SPR'*) and the duration of passage between predefined scheme entry and exit points. However, it is also extremely helpful to observe fish behaviour in the vicinity of screens and bywashes. Where problems occur or scheme passage rates are below desirable levels, this allows the modes of failure to be identified. A variety of methods can be used.

Table 7.2 lists methods that have been used in different applications and references literature describing the methods.

Table 7.2 Examples of methods used for fish screening system trials

(a) Fish sampling techniques

Technique	Application	Purpose	Reference
Intake/tailrace netting – using trawl-type nets	Sampling fish at water offtakes or hydropower plant	Estimation of numbers of fish passing inlet screen	Wood *et al*., 1994; Turnpenny *et al*., 1996; Hadderingh, & Smythe, 1997 Hadderingh & Bakker, 1998
Canadian screw-trap	Sampling fish in fast-flowing, deep channel	Estimation of numbers of fish passing inlet screen	Spiby, 2004
Electrofishing	Sampling areas behind screens	Estimation of numbers of fish passing inlet screen	-
Collection of fish from band- or drum-screens	Sampling fish at secondary (trash-) screening points, e.g. in power plant or water pumping stations	Estimate impingement rates; compare numbers impingement with/without behavioural technology	Turnpenny, 1993; Turnpenny *et al*, 1994, 1995; Maes *et al*., 2004
Louvre-screen trap	Sampling fish in major flow downstream of intake	Estimate entrainment rate; compare for different screening techniques	Solomon, 1992
Electronic fish counters	Intake and bypass flow counting of smolts and larger fish	Compare fish numbers in screened flow and bypass	Welton *et al*. 2002
Fish trapping with Wolf grid	Sampling fish in bypass channels	Estimation of numbers of fish diverted into bypass	Turnpenny *et al*., 2003b

(b) Fish observation and monitoring techniques

Technique	Application	Purpose	Reference
Video surveillance	Monitoring fish in front of screens & at bypass entrance	To check for evidence of fish impingement, delay or to observe behaviour	Larinier & Travade, 1999
Hydroacoustic monitoring	Intake forebays & bypass entrances	Observe behaviour & estimate relative numbers of fish passing screens or entering bywash	**7.3.2** Iverson, 1999; Johnson *et al*., 2002
Radio-or acoustic-tagging (biotelemetry)	Observing fish movement through or around schemes	Estimation of *SPR* and duration of passage	**7.3.3** Lariner & Travade, 1999
Passive Integrated Transponder (PIT) or Floy tagging	Monitoring fish movement through or around schemes	Estimation of *SPR* and duration of passage using mark-recapture	Larinier & Travade, 1999
Float-tagging	Observing fish movement through or around schemes	Detailed observation of fish behaviour in front of screens and bypasses; estimation of *SPR* and duration of passage	Turnpenny *et al*., 2003

Plate 7.1 Total flow netting at the draft tube of a small hydropower turbine. Here, fish enter a live-car at the cod-end of the net to facilitate removal and to reduce the likelihood of net-induced injuries .

7.3.4 Sampling Fish Post-Screening and in Bywashes

One of the most common methods involves fitting a net to filter the entire intake flow or some portion of it (Wood *et al*., 1994; Turnpenny *et al*., 1996; Hadderingh and Smythe, 1997 Hadderingh & Bakker, 1998). The net is often conical or pyramidal in shape, similar to a trawl net. The mesh size is selected according to the size of fish to be retained. The size of the net will depend on the flow rate and debris load. The force of water can be substantial and such nets should be generously sized and strongly attached. It is normally necessary to be able to shut off the flow to examine the net, but having the net open at the cod-end and discharging into a removable live-car (Plate 7.1) avoids this necessity and creates a more benign holding area for the fish. Alternatively, where space allows, fyke nets can be set in the flow behind the intake to capture all or a sample of the flow. With either approach, catches can be compared with and without the screening measures operating and with alternative screening arrangements.

A newer technique is provided by the rotary screw trap (Plate 7.2). The trap is fixed to a flotation raft and is water powered. The central drum rotates and lifts fish into a holding tank. The trap needs a water depth of ≥1m and works best in water velocities of ~1ms^{-1}. These conditions are usually met in hydroelectric headrace canals, where the trap has been used with some success to test a BAFF™ screen (Spiby, 2004). The screw trap samples only a small proportion of the total channel flow and therefore takes only a sample of fish.

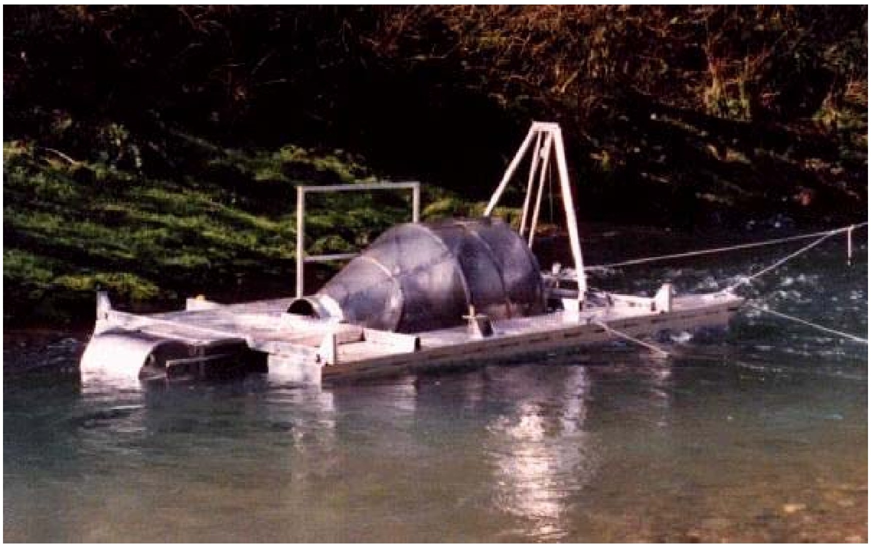

Plate 7.2 A rotary screw trap. Flow rotates the barrel and fish present in the flow are lifted into a holding chamber. The trap takes a small sample of the total channel flow.

Other kinds of fish trap are also useful. The Wolf trap (Plate 7.3) is essentially a dewatering device that can be placed in a bywash channel to remove fish. It comprises a slatted channel floor through which flow passes, leaving the fish to be washed along the surface of the slats and to drop into a tank or cage at the downstream end. It is probably the most widely used style of smolt trap and is ideally suited to monitoring fish entering a bywash (Turnpenny *et al*., 2003b). The louvre screen trap described in section 3 provides a further alternative which appeared to work well, although the version constructed at Walton was a substantial structure that would be more suited to a semi-permanent test facility.

Plate 7.3 A Wolf trap photographed at Halsou, France. Water is strained through a set of parallel rods or slats that run along the length of the channel. These are spaced at about 6mm apart, allowing water to pass through, while fish are flushed along the slats and into a holding tank at the downstream end.

Electrofishing areas downstream of a fish screen can provide useful information about fish 'leakage' through the screen. It is not normally used as a quantitative method, as fish may quickly pass beyond reach of the gear, e.g. into culverts or pipes. The presence of fish in quiet areas behind a screen can signal problems with screens.

Sampling on band and drum screens is the most widely used method for assessing entrainment rates at thermal power station cooling water intakes (Turnpenny, 1993; Turnpenny *et al*, 1994, 1995; Maes *et al*., 2004) and potable water intakes (Frear and Axford, 1991). This allows catches to be compared e.g. for 'on' and 'off' periods with acoustic deterrents, bubble curtains strobe lights, etc.. Care must be taken to allow for any transit time of fish through the system or residence time before fish appear on the screens, which can range from minutes to many hours. Transit and residence time can be estimated e.g. by introducing marked fish into the intake and timing their arrival on the screens, but it may vary according to the swimming abilities of the fish (influenced by species, size, water temperature and other factors) and their behaviours. Once this is known, a suitable gap can be left between test conditions.

Welton *et al*. (2002) devised automated smolt counting methods to simultaneously count fish passing with the screened flow and in the bywash flow during trials of a BAFF™ and bubble curtain system. Counting was performed by crowding the fish into a narrow channel fitted with resistivity counters (Plate 7.4). Time-lapse video recording was used to validate the counters. The relative numbers of fish entering the bypass versus those in the main flow were used to assess screening efficiency.

Plate 7.4 Video and resistivity smolt counting unit used to monitor smolt diverted into a bywash during BAFF™ tests conducted by the Centre for Ecology and Hydrology (CEH) at the River Frome test site in Dorset.

7.3.5 Observing Fish Behaviour

Table 7.2(b) lists methods commonly used for observing fish behaviour around screens and bywashes. While 'hi-tech' solutions such as radio-tracking and hydroacoustics have their place, the value of simple direct observation should not be overlooked. On many schemes, much can be learnt from a few hours personal observation at key points such as along the screen face and at the bywash entrance. Arrays of overhead or underwater CCTV cameras strategically placed can provide much information about how fish react to the screens or behavioural barriers and where points of weakness are. It may be possible to link these to e.g. hydraulic anomalies at those points, allowing corrective action to be diagnosed. As visibility can often be poor, visualisation of fish movements can be aided by fitting brightly coloured or luminous tags to the dorsal fin of the fish. Alternatively, float tags can be used. Turnpenny *et al*. (2003) describes their use in testing a BAFF™ at Hemsjo Nedre hydropower plant in Sweden. For smolt-sized fish (i.e. 12-20cm in length), 12mm diameter polystyrene beads attached by a 0.5m length of fine nylon line are suitable (Plate 7.5). These can be sprayed with fluorescent paint to aid visibility. The tags should be attached very lightly, near to the edge of the dorsal fin membrane, so that the fish can easily pull away from the line if it should become snagged. In a recent performance trial of the angled bar rack and bywash at Stanley Mills hydropower plant in Perthshire (Dr S.C. Clough, personal communication), float tags attached to the smolts were also fitted with miniature chemical lights (Starlite™, Luminasa Europe Ltd). Using this technique, it was possible to observe the movements of smolts released into the headrace, plotting their paths and timing their rate of passage into the bywash. It was demonstrated that Atlantic salmon smolts reacted positively to strong water currents, entering the fastest flowing areas and avoiding contact with the screens.

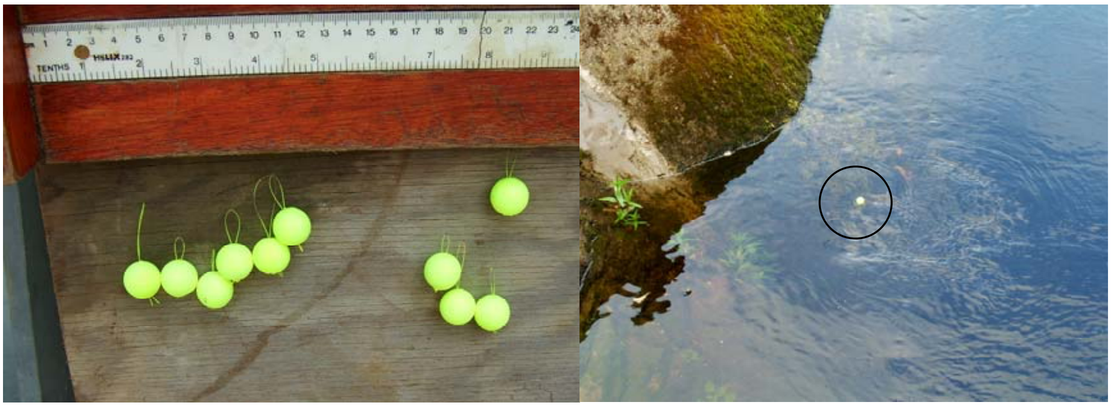

Plate 7.5 Fluorescent polystyrene float tags as used to follow close-range movements in front of screens and bywashes. The left-hand photograph shows a number of floats prior to attachment; the right hand side shows one as viewed in the water (inside black circle).

Hydroacoustic methods have been used extensively in the USA to observe detailed movements of fish at large dams (e.g. Iverson, 1999; Johnson *et al*., 2002) but these methods are costly and somewhat experimental and to date have not been used routinely in the UK.

Radio- and acoustic tracking provide useful techniques for making observations on a larger scale, i.e. to monitor fish passage of an entire scheme or network of schemes (Turnpenny *et al*., 1996), rather than fine detail close to screens and bywashes, although quite fine resolutions (± a metre or so) can be achieved in some circumstances (Larinier & Travade, 1999). Numbers of fish that can be economically used with these methods may be limited, in which case they may be augmented by cheaper tagging methods using PIT or Floy tags. The latter types are normally used in mark-recapture programmes, results being derived e.g. from the numbers of recaptures obtained in bywash and discharge net samples from a known number of tagged fish released (Larinier & Travade, 1999).

7.4 Site-Specific Commissioning Trials

Generally it is advisable to conduct at least a brief assessment of the performance of a screening system following installation or modification. This should be scheduled as part of the commissioning process. It is highly desirable to consider safe access and sampling requirements at the design and construction to facilitate future monitoring. Regulatory agencies may require provision to be made for fish trapping or counting in bywash channels. For example, following consultations with the Environment Agency, a recent planning application for construction of a small hydropower plant on the Yorkshire

Ouse includes provision of a removable Wolf-grid in the bywash channel and a fish holding chamber. At another site where commissioning trials were required, the failure to provide safe access for sampling proved costly when extensive temporary scaffolding was needed.

Trial methods for site-specific commissioning may be selected from any of those identified in section 7.3 but for cost reasons the simpler methods are to be preferred., e.g. trapping, video surveillance or float-tagging. For regulated applications the methods will need to be agreed with the appropriate agency in advance.

8 KNOWLEDGE GAPS AND FUTURE RESEARCH NEEDS

8.1 Review of recommendations from Solomon (1992)

In advice to the National Rivers Authority, Solomon (1992) made a number of recommendations in respect of fish screening. These are summarised below, along with an indication of progress made since 1992 and comments on their current relevance. The words shown in quotation marks are paraphrased from Solomon's.

8.1.1 *"A national database on abstractions should be developed to include details of fish protection stipulations and measures actually fitted."*

A national database of some 48,000 abstractions now exists but details of fish protection stipulations and measures are not included. <u>This remains a need.</u>

8.1.2 *"Staff should be provided with a concise legal summary of legislation pertinent to fish screening and enforcement and that there should be a broadening of existing legislation to include all types of abstraction and all species of fish."*

Following formation of the Environment Agency in 1995, powers under SFFA 1975 were transferred to the Agency and a national training programme on fish screening techniques and legislation for enforcement staff was undertaken. This Guide provides further information on the subject for the benefit of regulatory agencies and the public.

Powers to extend the scope of screening as proposed have been recommended in the recent Fisheries Legislative Review. At present they have not been enacted in law. However, a review of SFFA s.14 & 15 under the Environment Act 1995 extended its power to include fish farm intakes and outfalls.

8.1.3 *"Operators should be required to fit appropriate fish screens whenever possible on new and existing abstractions, subject to provisions of the law."*

Since 1995, the Agency has carried out a national SFFA s.14 enforcement programme at Regional and Area levels. Many intakes that were formerly not provided with fish protection screening have now been retrofitted with screens or alternative measures (e.g. behavioural technologies), including some that were not legally obliged to do so. This has been undertaken following the Agency's Risk Assessment approach detailed in section 4.2. See also point 8.1.8 below. In England and Wales, provision of fish appropriate screening is also now addressed through conditions attached to abstraction and impoundment licences under the Water Resources Act.

8.1.4 *"R & D should be commissioned to investigate the timing, mechanisms and extent of migrations of 0+ and older coarse fish to assist in better defining periods when abstraction might be stopped."*

The Agency has commissioned work on coarse fish migrations (see review by Lucas *et al*, 1998: EA Technical Report W152). Other UK research has also since contributed to this field (see e.g. Smith, 1998; Turnpenny *et al*, 1998b). However, while knowledge in this area has improved, regional and between-river differences in seasonal timing are not well known and further research into geographic differences would help to refine operating agreements. There will nevertheless remain a need to retain flexibility in such agreements to allow for inter-annual variations.

8.1.5 *"R & D should be commissioned to investigate distribution and dispersion dynamics of coarse fish to aid in sympathetic siting of intakes (including diurnal patterns, swimming depths, etc.)."*

<u>No significant advance has been made in this field and further research is required.</u>

8.1.6 *"R & D should be commissioned to investigate population control mechanisms in 0+ fish to assess impact of losses at various life-stages."*

Work by Smith (1998) and Turnpenny (1999) used the Equivalent Adult Value (EAV) concept (section 2.2), which enables comparisons of impacts of various life-stages but omits any density-dependent population control effects.

Given the inherent difficulties in quantifying the compensatory capacity associated with density-dependent effects (Van Winkle, 1977), it is probably more useful to assume that no compensation takes place and act accordingly. The Equivalent Adult approach is an effective tool for this purpose. While this has been used by Smith (1998) and Turnpenny (1989) to make some attempt at quantifying population level impacts, both authors were aware that the outputs are only as good as the life-history data entered into the analysis. Such data (age-specific mortality and reproductive rates, sex ratios) are presently sparse and are likely to be somewhat specific to particular river conditions, fish communities and even years. <u>The most pressing research need in this context is therefore to assemble life-history data sets for particular species. This might best be done in the first instance by investigating benchmark communities representative of key habitat types (lakes, upland streams and rivers, lowland streams and rivers etc.). The benefits of such research would no doubt spill over into other areas of fisheries biology and management and therefore might be funded from multiple sources.</u>

8.1.7 *"R & D should be commissioned to investigate screen slot and mesh sizes suitable for different species and lifestages."*

Earlier work reported by Turnpenny (1981) is considered to provide an adequate basis for calculating mesh size for the majority of species, based on the body-length/body-diameter relationship ('fineness ratio'). Further work is however needed to clarify the more complex relationship between slot-width, channel velocity, fish body length and exclusion efficiency for PWWC screens. <u>This area is currently being researched in the USA and any new data from those studies should be investigated before commissioning new UK work. As the PWWC screen is one of the most important screening techniques currently available, good information on these aspects is essential and work should be commissioned if data are not found elsewhere.</u>

8.1.8 *"R & D should be commissioned to investigate the extent of fish entrapment at intakes in England and Wales."*

Some work in this field, described in section 2 of this Guide, has been undertaken since Solomon's (1992) recommendations, particularly pertaining to thermal and hydropower generating plant. <u>As the main large-volume water abstractors, there remains a need to investigate potential impacts from power plant abstractions, either through commissioned R & D</u>

or owner-funded studies. The latter currently account for most of the studies carried out in this field, usually as a result of conditions attached to operating licences but in some cases volunteered by the owners. Future work should concentrate in particular on designated fish species (see section 1.2), especially lampreys, on entrainment of fish eggs, larvae and fry that are usually not fully represented in power station sampling and on other species of conservation interest such as sea trout, smelt and eel.

The little work that has been carried out at potable water and fish farm abstractions (e.g. at Walton on Thames and the Hampshire Avon – section 2) suggests that impacts on coarse fish through fry entrainment are potentially large and significant (Turnpenny, 1999), although the importance of high entrainment counts is hard to judge in the absence of a clear understanding of the population dynamics. All evidence indicates that redistribution movements of fry in rivers put large numbers at risk of entrainment and the absence of data on fry movements in particular water bodies should not be used as an excuse for not taking adequate screening measures to protect fry. Wherever possible, through legislative provisions or voluntary cooperation, owners should be encouraged to ensure protection of all life stages of fish. This may be best achieved through screening measures, or through temporal modulation of flow to avoid abstraction during periods of high entrainment risk.

8.2 Additional Recommendations for R & D

The need for adequate site trials of different existing screening technologies was addressed in section 7, where the purpose of the proposed studies was to test the suitability and efficiency of different techniques for the variety of species and applications of interest. The aim here is to consider particular gaps in either our understanding of impacts or the armory of available techniques not covered in previous discussion. Some are identified below. Undoubtedly others will become apparent when looking into individual project applications.

8.2.1 Juvenile fish mortalities and injuries in hydroelectric turbines

Section 4 refers to techniques used to quantify fish mortalities in turbines. Numerical modelling techniques have been validated on fish of smolt size and larger and appear to provide adequate predictions of mortality rates for impact assessment purposes (Turnpenny *et al.*, 1998, 2000). Theory and empirical laboratory evidence (Turnpenny, 1988) indicate that mortality rates due to blade strike - the major cause of injuries in smolts- are relatively much lower in small fish (<20g) than for smolt-sized fish. This results from the larger surface area-to-mass ratio in small fish, which causes the water to drag them around the blade's leading edge, whereas the momentum of larger fish makes collision more probable. The present STRIKER turbine fish injury model developed by Turnpenny *et al.* (2000) does include terms to account for pressure flux and turbulence (shear stress) effects, which may also contribute to fish injury risk, but the empirical data used were derived for smolt-sized and larger fish, therefore they may be unreliable in application to juveniles.

At present, no lowland river hydropower schemes in England and Wales are known to provide screening of sufficiently small mesh opening to prevent the entry of coarse fish fry. For low-head, run-of river schemes the inherent hydraulic head-loss caused by low porosity screens make them impractical and uneconomic. From theory it has been assumed that fry mortalities would be negligible. There is now an urgent need to test this assumption through monitoring at operational lowland river sites and to modify predictive models as necessary.

8.2.2 Eel and lamprey screening and guidance methods

These elongate, thin-bodied fish are poor swimmers and present particular problems with regard to fish screening. Where it is practicable to use fine-meshed or wedge-wire screens, it has been shown that both species can be effectively screened down to juvenile sizes. At large abstractions, especially at thermal and hydropower generation sites, small aperture screens are not always practicable; these species are particularly at risk of blade strike during passage through hydroelectric turbines. While a number of behavioural guidance techniques have shown promise, none is currently sufficiently developed to recommend.

The main contenders for behavioural guidance of these species at present are:

- strobe lights
- low-frequency sound
- water currents
- electric fields.

All of these methods should be investigated, individually and in combination. Juvenile and adult eels and lampreys should be included in tests. Tests would best be carried out in the headrace of one or more small hydropower stations where hydraulic conditions are relatively well controlled. A number of suitable sites have been identified.

One further technique of interest is the bed-level eel bypass channel such as installed at Backbarrow hydropower scheme on the R. Leven in Cumbria and elsewhere in Europe (see section 3.5.3). It is not clear how effective this device is, nor whether it could be improved with better hydraulic conditions but available evidence suggests that it is at least partially effective under some flow conditions. This method merits further consideration as an adjunct to other screening and guidance methods. In particular, it will be necessary to observe the detailed behaviour of eels confronted with this type of bypass and to be able to manipulate the dimensions and geometry of the structure. Such an approach would lend itself to testing in a laboratory flume or raceway, where the behaviour of eels and lampreys could be observed.

8.2.3 Behavioural Barriers in General

Owing to the potential cost savings and ease of use, there is considerable interest from operators in using behavioural barriers as an alternative to positive exclusion fish screens for a wider range of species. This is particularly true of low-head run-of-river hydropower schemes, which use large volumes of water and where screening costs are comparatively high and hydraulic losses may limit generating revenues (see Turnpenny *et al.*, 1998a). Although behavioural techniques have advanced, scientific trial results are still limited and regulatory agencies are reluctant to accept their use in many situations. The cost of conducting scientific trials to a sufficiently rigorous standard can be too high for individual operators of small schemes to bear and therefore further generic testing is recommended. Sites of existing or proposed schemes on the rivers Trent (e.g. Beeston), Yorkshire Ouse (e.g. Linton), Thames (various) and some Scottish rivers may provide suitable locations.

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10 GLOSSARY

The following are definitions of certain words and abbreviations used in this Guide:

AFD – 'Acoustic fish deterrent system' – a behavioural screen or barrier exploiting the hearing sensitivity of fish.

Acoustic barrier – Barriers which exploit the hearing sensitivity of fish.

Approach velocity – The velocity of water approaching the screen.

Attraction flow – A water flow which attracts fish to a desired area.

Backflush – Reverse of flow to wash off debris from the screen.

BAFF – 'Bioacoustic fish fence' – a combined sound and bubble curtain screening system.

Behavioural barrier or screen – A fish deterrent system which works by stimulating the senses of fish either by repulsion or attraction mechanisms.

Benthic – Bottom dwelling fish.

Biofouling – the build up of aquatic organisms on a substrate.

Bubble curtain or barrier – A wall of bubbles used to deflect or guide fish.

Bypass – A channel or pipe which allows fish to pass by the obstruction unharmed via an alternative route.

Bywash – Synonymous with 'bypass' (see above) but more commonly used in Britain

Channel velocity – The velocity in front of the screen measured axial to the flow channel.

Coanda effect – Principle of how fluids follow a surface identified by Henri Coanda in 1910.

Diadromous – Migratory species that move between the sea and freshwater and vice versa.

Epibenthic – Species that normally live close to the bed.

Escape velocity – The water velocity perpendicular to the face of the screen.

Entrainment – The drawing-in of fish of any lifestage at a water intake

Euryhaline – Species with a wide tolerance of salinities

GFFB™ – 'Graduated field fish barrier' – a form of electric screen which presents an electric field of increasing intensity (voltage) as the fish gets closer, generated by means a series of separate pulse generators

Impingement – The accidental pinning of fish onto the surface of a screen by the water current

Infrasound – Sound with a frequency of less than 20Hz

Kelt – Stage in a salmon life cycle just after spawning.

Lithophilous – Requiring gravel on which to spawn.

MLES – 'Marine life exclusion system' – a water-permeable geotextile barrier.

MSSS –'Maximum sustainable swimming speed'

Phototaxis – Movement in relation to light.

Pinhead fry – Newly hatched fry.

PWWC – Passive wedge wire cylinder – a type of fine aperture screen suitable for fish exclusion down to fry size

Retrofit –Addition of equipment to existing facilities.

Rheotactic – Movement (of fish or other animal) in relation to flow.

SAC – 'Special area of conservation'.

Shear (hydraulic) – Differential velocity field in water

Smolt – Young salmon of 2 or 3 years old.

SPA – 'Sound projector array' – uses arrays of underwater transducers to produce a diffuse field of sound.

SSSI – Site of Special Scientific Interest

Strobe light – High intensity, short duration light pulses.

Teleost – A bony fish.

Transformer – Recently metamorphosed pre-adult lamprey

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Environment Agency

Cooling Water Options for the New Generation of Nuclear Power Stations in the UK

SC070015/SR3

Better regulation programme

Evidence Directorate

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[redacted]

Miranda Kavanagh

Director of Evidence

Executive summary

The consideration of new nuclear power stations is split into two phases. The first phase addresses generic design matters (namely, acceptability of candidate nuclear power station designs) and the second deals with site-specific applications for authorisations under the Radioactive Substances Act 1993.

The Environment Agency, Health and Safety Executive's Nuclear Installations Inspectorate and the Office for Civil Nuclear Security are currently assessing candidate designs of new nuclear power stations. The Environment Agency is exploring the environmental effects of candidate designs based on a generic site description. A statement about the acceptability of that design for a generic site in England and Wales will be provided.

Cooling water is required to remove "waste heat" from power stations regardless of whether the stations are nuclear or conventional. A nuclear power station has a typical thermal efficiency of 25-33% (compared to around 40% for a modern coal-fired station) and hence a 1,000 megawatt electric (MWe) nuclear station would typically generate up to 2,000 megawatts of low-grade waste heat. The reasons for this apparent wastage are explained. The report also explores cooling water options for new reactors and evaluates their potential environmental impacts in terms of effects on biota, and thermal, chemical and radionuclide pollution. The findings are focused on, but not confined to, nuclear plants and will have general applicability to other large (above 1,000 MWe) thermal power station projects.

This report is based on publicly available information and publications and the experience of the authors. It provides an overview of power station cooling water systems in use in the UK and abroad. Details of cooling water options for new nuclear power stations in the UK are given. Cooling water system design (direct and indirect cooling water systems, intake and outfall designs), how the design affects the performance of the cooling option and issues such as temperature differentials between water intake and discharge are discussed.

An overview of environmental issues associated with cooling water systems of nuclear and other large power stations is presented. Issues arising from water abstraction and discharges are discussed. These include fish and invertebrate intake and impingement on filter screens, effects of passage of planktonic and small life-stages through the cooling system, thermal, chemical, radionuclide pollution, and effects of cooling tower emissions to air. The report also discusses different mitigation measures (such as intake location, intake screen designs to minimise impacts of entrapment, entrainment and impingement, plume abatement techniques to minimise effects of plume formation). Implications of combining conventional liquid discharges within the cooling water discharge are also considered.

Environmental issues with specific cooling water options are identified and explained. The report evaluates cooling water options in terms of environmental concerns (including water demand and energy efficiency) and assesses the best options for use in different types of water environments (coastal, estuarine and fresh waters). Effects of climate change on the choice of cooling water options are also briefly considered.

It is likely that new UK nuclear stations will be built on coasts or estuaries. A key question at the outset of the study was whether direct cooling (also known as "once-through") can still be considered Best Available Technology (BAT) for large coastal and estuarine power stations, as set out in the European Commission's BAT reference document on industrial cooling systems (BREF-Cooling, 2001). While this has recently been challenged in relation to a proposed 2,000 MWe combined cycle gas turbine (CCGT) power station at Pembroke, the findings of our study indicate that direct cooling can be BAT for estuarine and coastal sites, provided that best practice in planning, design, mitigation and compensation are followed. The potential BAT-status of direct cooling has essentially been preserved owing to improved understanding of survivability of the entrainment process, and substantial developments in impingement mitigation techniques since the BREF was written. As per the BREF advice, there may remain cases where, even with the application of best practice, residual impacts would be unacceptable. In these cases, seawater cooling towers would be used. BREF advocates the use of dry-cooling methods only where water is in extreme short supply; this advice remains appropriate. These conclusions are generic and site specific applications will be assessed individually.

The findings are applicable to both nuclear and conventional power stations. We conclude that direct cooling may be the best option for some nuclear power stations. A summary of impacts from the various cooling is summarised below:

Environmental concern	**Direct cooling**	**Cooling towers**		
		Natural draught (wet)	**Mechanical draught (wet)**	**Natural draught (dry)***
Generation efficiency	High efficiency Uses less fuel so lower aerial emissions	Typically 0.5 - 1.5% less efficient than direct cooling	Typically ~2% less efficient than direct cooling	Lowest efficiency 2 - 3% less efficient than direct cooling
Complexity	Low	Moderate	High	Very high
Water abstraction	High	Moderate/low	Moderate/low	None
Abstraction effects	Site-specific -depends on characteristics of receiving waters			
Water consumption	None on-site	Moderate	Moderate	None
Visible plumes	None	Moderate	Moderate/low	None
Ground fog & icing	No icing. Local fog plume over shoreline discharges	None	Possible	None
Visual impact	Occasional foam or 'slick' at outfall	High	Moderate	High
Noise	None	Low	Moderate	Low/none
Discharge effects	Site-specific -depends on characteristics of receiving waters			
Waste disposal to landfill**	None if using fish recovery & return***	Moderate	Moderate	Moderate/none
Land use on-site****	None/low	Moderate/high	Moderate	High

* See sections 3.1.9 and 3.1.11

** Wastes from wet towers are mainly silt (non-hazardous); from dry towers, glycol (non-hazardous), if used

*** See section 7.2.3 'Consenting Issues' and section 6.1.6 'Biota recovery and return techniques'

**** This covers buildings and structures only and does not include spray ponds or cooling canals

Contents

1	**Introduction**	**1**
1.1	Purpose of study	1
1.2	Background	1
1.3	Terms of reference	3
1.4	Sources of Information	3
2	**Why power stations need cooling**	**4**
2.1	Thermodynamics and the steam cycle	4
2.2	Improving efficiency	6
2.3	The role of the condenser	10
2.4	Principles of cooling	11
2.5	Economics	13
3	**Existing power station cooling systems**	**15**
3.1	Critical review and description of alternative cooling circuits	15
3.2	Issues on the use of seawater in cooling towers	34
3.3	Reactor cooling and ultimate heat sinks	37
4	**CW system design**	**38**
4.1	Direct CW systems	38
4.2	Indirect CW Systems	48
4.3	Choice of CW System	53
5	**CW system design for new UK nuclear power stations**	**55**
5.1	Introduction	55
5.2	Potential sites and water sources	55
5.3	Basis of design	56
5.4	CW intake design	56
5.5	CW intake screening	60
5.6	Biofouling control	73
5.7	Essential cooling water supply	76
5.8	CW outfall design	77
5.9	Heat dispersion	79
5.10	Liquid radioactive waste disposals with the CW discharge	81
6	**Environmental issues associated with cooling**	**83**
6.1	Effects associated with abstraction	83
6.2	Effects of cooling towers	123
6.3	Effects of the thermal discharge	128
6.4	Using the cooling water stream to amend conventional discharges	165

7	**Conclusions and recommendations**	**166**
7.1	Environmental effects of cooling systems	166
7.2	CW system design best practice	170
7.3	Status of BAT definitions for cooling water systems	182
7.4	Dealing with residual impacts	184
8	**References**	**185**
Glossary & list of abbreviations		**194**
Appendices		

List of tables and figures

Table 2-1	Improvements in theoretical efficiency and in performance, using a constant 35°C turbine exhaust temperature	7
Table 2-2	Increasing production (kilowatt-hours) from one tonne of coal	7
Table 3-1	Cooling capacity of some reservoirs. For purposes of comparison nuclear stations are assumed to be 33%, and conventional stations 40%, thermally efficient.	20
Table 3-2	Cooling capacity of Turkey Point (Fla) canal system	22
Table 3-3	Relative areas required for cooling using different methods	23
Table 3-4	Relationship between concentration factor, half-life (retention) and make-up rate, assuming one per cent evaporative loss of the circulating volume	27
Table 3-5	Economics of four methods of evaporative enhancement of dry coolers (adapted from Kutscher and Costenaro, 2002)	32
Table 5-1	Electrical out put and CW demand for the EPR and AP1000 reactors	55
Table 5-2	Detailed design of submerged offshore intake structure	58
Table 6-1	Typical fish survival reported from studies of drum or band screens with simple modifications for fish return (Turnpenny and O'Keeffe, 2005).	92
Table 6-2	Percentage entrainment mortalities of a range of planktonic species under "normal" power station levels of stressors, interpolated from series of EMU experiments (data from Bamber & Seaby, 1993; 1995a,b; 2004; Bamber *et al*. 1994).	100
Table 6-3	Sizewell A Power Station, 1981-82 Study. Estimated annual loss to the fishery of commercial-sized fish due to CW abstraction (after Turnpenny *et al.* 1988)	101
Table 6-4	Equivalent adult tonnages of key commercial species impinged at four UK estuarine power stations, compared with reported England & Wales commercial landings from adjacent sea areas (Turnpenny, 1988b).	103
Table 6-5	California Energy Commission figures for habitat areas that would be required to replace entrainment losses at Californian power plants (CEC, 2005). Plant CW flow shown for comparison.	104
Table 6-6	Deflection efficiencies reported for the acoustic fish deflection system at Doel nuclear station (Maes *et al*. 2004)	115
Table 6-7	Example of large-scale habitat compensation projects in the UK (Scottish Parliament, 2008)	122
Table 6-8	Draft WFD standards against requirements for transitional waters to have good ecological status	131
Table 6-9	Bioaccumulation potential of some chlorine produced byproducts (based on Chemical Database Management System, Chemwatch Package 2004/1)	136
Table 6-10	Effect of temperature and salinity on DO solubility (from Turnpenny, Coughlan and Liney, 2006)	137
Table 6-11	Draft WFD dissolved oxygen standards for transitional and coastal waters of different ecological status (UKTAG, 2008)	137
Table 6-12	Simple mass-balance equations for discharges to various water environments (Environment Agency, Environment and Heritage Service, Scottish Environment Protection Agency, 2003; Environment Agency, 2008a; Environment Agency 2009a)	142
Table 6-13	Key model input parameters	146
Table 6-14	Experimental data on thermal avoidance thresholds of juvenile estuarine fish (Jacobs, 2008)	158
Table 7-1	Potential loss of electrical output through the use of some alternative methods of cooling a 1,000 MWe power station, using direct cooling as the base case (Mclauchlan, 2009).	170
Table 7-2	A comparison of cooling options	171
Table 7-3	Turbine backpressure and steam condensation temperature (saturation value) for a range of nominal CW intake temperatures (10ºC ΔT)	181

Figure 2-1	Internal view of a 500 MWe turbine. Small-diameter HP cylinder is in the foreground, then an intermediate pressure cylinder and three LP stages. Overall length of unit is 55 metres and large LP disc (blades) diameter is 2.5 metres.	5
Figure 3-1	**Oldbury tidal reservoir: impounding wall is above level of falling tide**	18
Figure 3-2	Aerial view of Turkey Point cooling canals	22
Figure 3-3	**Spray cooling (photo courtesy Siemens Power Generation)**	**24**
Figure 4-1	Diagram of typical direct CW system	38
Figure 4-2	Inshore Intake at Ballylumford B, Larne	41
Figure 4-3	Inshore Intake at Kilroot, Carrickfergus	41
Figure 4-4	Offshore intake and inshore outfall at Wylfa, Anglesey	42
Figure 4-5	Capped radial flow intake structures for Vasilikos, Cyprus	42
Figure 4-6	Bolted tunnel and shaft lining for Sizewell A	44
Figure 4-7	Precast offshore outfall conduits for Kilroot	44
Figure 4-8	CW pumphouse substructure under construction at Vasilikos, Cyprus	45
Figure 4-9	Inshore outfall structures for Heysham A and Heysham B	47
Figure 4-10	Model of submerged outfall for Kilroot	48
Figure 4-11	Diagram of typical indirect CW system	49
Figure 4-12	Configurations of natural draught cooling tower	50
Figure 4-13	Configurations of induced draught cooling tower	50
Figure 4-14	Typical hybrid tower arrangement	51
Figure 4-15	Diagram of typical direct CW system with 'helper' tower	54
Figure 5-1	Typical submerged intake structure	57
Figure 5-2	Temporary works for immersed tube construction, South Humber	60
Figure 5-3	CW intake at Keadby, showing overhead gantry rail for screen raking system and accumulated brushwood	63

Figure 5-4 Top: passive wedge-wire cylinder screen in 'T'-format. An internal core-tube with holes of varying diameters is used to obtain an even flow distribution across the screen surface (courtesy Hendrick Screens). Bottom: PWWC screen array for bankside use. 64
Figure 5-5 Engineering drawings of travelling band screen: vertical section (left) and elevation (right) 66
Figure 5-6 Characteristics and flow arrangements for travelling band screens in plan view: a) through-flow, b) dual flow (UK and France), c) central flow (Germany) 67
Figure 5-7 Schematic showing in-to-out (UK) versus out-to-in (France) drum screen design concepts 68
Figure 5-8 Further comparison of out-to-in and in-to-out drum screen designs 70
Figure 5-9 Overhead view of typical set of four drum screens at a coastal plant 72
Figure 5-10 Inshore outfall at Vasilikos, Cyprus 78
Figure 5-11 Layout of Sizewell A and Sizewell B 80
Figure 5-12 Temperature drop with distance from discharge point for Sizewell B 81
Figure 6-1 Mesh size curves for screening fish of different body shapes (from Turnpenny and O'Keeffe, 2005, after Turnpenny, 1981) 85
Figure 6-2 Bandscreen panel fitted with six-mm square woven stainless-steel mesh. The trash elevator is in the form of a narrow ledge which does not retain water and is unlikely to properly retain larger or writhing fish such as eels. 86
Figure 6-3 Sorted sample of fish collected from power station screens (left). Catches at saltwater sites typically contain a wide range of species but are often dominated by pelagics, such as sprat and herring. The bulk of the catch is often made up of juveniles or smaller species of under 20 cm in length, although certain locations and intake designs can put larger fish at risk (right). 87
Figure 6-4 Coarse screen blocked by fouling 88
Figure 6-5 Estimated annual total quantities of fish impinged at UK estuarine and coastal power stations (Turnpenny and Coughlan, 1992) 88
Figure 6-6 Examples of fish injuries caused by impingement. Left: bleeding into the eye is common as a result of exposure to pressure change. Right: an eel with spinal fracture. 91
Figure 6-7 Effect of chlorine concentration on carbon fixation by phytoplankton at Fawley Power Station, Hampshire (Source: Davis, 1983) 95
Figure 6-8 Percentage mortality of adult calanoid copepods within one hour of entrainment under various chlorination regimes at four different UK power stations (Source: Coughlan & Davis, 1983) 97
Figure 6-9 Schematic of FARL Entrainment Mimic Unit (EMU) (after Bamber *et al.* 1994) 99
Figure 6-10 Equivalent adult value curves for common UK commercial species (Turnpenny, 1989) 102
Figure 6-11 Area of replacement habitat for entrainment losses against flow rate (plotted from data in Table 6-5) 105
Figure 6-12 Fawley Power Station CW intake at low tide, showing the onshore CW inlet channel and bordering saltmarsh areas 107
Figure 6-13 Cefas Young Fish Survey data showing zonation of sole (*Solea solea)* relative to water depth 108
Figure 6-14 Schematic showing rising bubble plume and surface currents normal to the barrier line in a bubble curtain 110
Figure 6-15 Schematic showing bubble curtain deflection concept for an offshore intake 110
Figure 6-16 Velocity cap: (a) section of uncapped intake showing vertical draw-down pattern, (b) section of capped intake showing horizontal flow pattern, (c) as (b) but showing critical relationship between vertical opening [x] and length of horizontal entrance [1.5x] for fish reactions (after Schuler and Larson, 1975) 112
Figure 6-17 Concept design for a low-velocity, side-entry offshore intake structure with velocity cap, based on physical and hydraulic model tests carried out at Fawley Aquatic Research Laboratories (Turnpenny, 1988a and unpublished). A and B are the cylindrical caissons. Water enters through C. 113
Figure 6-18 Schematic layout of a sound projector array (SPA) acoustic fish deterrent system with six sound projectors. Large systems may use up to 60 sound projectors, each with a dedicated amplifier and diagnostics unit. 115
Figure 6-19 Example of an acoustic fish deterrent support structure constructed across a shoreline CW intake at the Lambton Generating Station, St Clairs River, Canada. The vertical beams each act as rails, which allow sound projectors to be raised and lowered for maintenance. The positioning of the structure ahead of the intake opening ensures that the fish repulsion zone is in an area of low water velocities from which the fish can escape. The inset shows an individual sound projector. (Main photo courtesy Paul Patrick, Kinectrics). 116
Figure 6-20 Example of fish return launders. Launders should be covered to reduce predation risk. Larger radius (3 m) swept bends reduce the risk of debris and fish becoming caught in bends. 117
Figure 6-21 Streamlines around a travelling screen fish bucket (a) traditional, showing rotational flow and (b) modified to stall flow within bucket (Fletcher *et al.* 1988). Width of bucket opening is typically 60 mm in standard designs. 118
Figure 6-22 Example of a fish bucket profile. The mesh to the left represents part of one of the main screen panels (Eimco Water Technologies). 118
Figure 6-23 Fish that have overshot the hopper collecting on a scaffold boards inside a drum screen chamber 120
Figure 6-24 A thermal plume, made visible by alginate-induced foaming 129
Figure 6-25 Schematic vertical section through an offshore thermal plume showing the hydraulic processes associated with dispersal. Close to the plume, where ΔT is highest, the water rises above the bed, so effects on benthos are usually small here. Further away, the plume mixes with the receiving water and rapidly cools. 130
Figure 6-26 Temperature preferenda, optimal growth temperatures and lethal temperatures for key UK fish species. Current UK water temperature standards are indicated by dashed vertical black lines. Suggested WFD boundaries are shown by dashed vertical coloured lines: green, high/good; blue, good/mod; orange, mod/poor. 150
Figure 6-27 Mortality of oysters in the River Blackwater, Essex, following the severe winter of 1962-63 (Turnpenny and Coughlan, 2003) 151
Figure 6-28 Percentage cover of Balanus balanoides (means with ranges; five replicates per station) for a transect of 10 sampling stations on the rocks leading away from Wylfa Power Station CW discharge (from Bamber, 1989). 153

Figure 6-29	Monthly densities of A – Tubificoides amplivasatus and B – T. benedii in the CW discharge canal of Kingsnorth Power Station, at six sampling sites reflecting a gradient of mean ΔT from 9.2ºC at 5 to 0ºC at DC4 (after Bamber and Spencer, 1984)	154
Figure 6-30	Continuous seabed temperatures in Kingsnorth Power Station CW discharge canal over two weeks in December 1980, starting at high tide, for sampling sites DC1 (one km from the outfall), DC2 (1.7 km) and the control site DC4 (four km)	155
Figure 6-31	Numbers of benthic infaunal species in Kingsnorth Power Station CW discharge canal over three years; sampling sites K1 and DC1 to DC3 are within the canal, DC4 is the control site (after Bamber and Spencer, 1984)	156
Figure 6-32	Growth curves for *Urothoe brevicornis* for the 1967 cohorts at Hunterston Power Station beach (solid circles) and the control beach at Millport (open circles) (redrawn after Barnett, 1971)	159
Figure 6-33	Growth curves for *Cyathura carinata* populations from Kingsnorth Power Station discharge canal (solid circles), compared with populations from Arcachon (open circles), the Kiel Canal (solid squares) and the Polish Baltic (open squares) (after Bamber, 1985); horizontal bars indicate winters	160
Figure 6-34	Example of a temperature exposure time-series for a specific habitat extracted from the Delft 3D model, comparing surface and bed temperatures	162
Figure 6-35	Modelled cross-section of a thermal plume entering an estuary channel, extracted from the CFX-5 model. This shows the buoyant plume rising from the outfall structure (to the left). Note that most of the channel cross-section contains ΔT values of under +3^{o}C, which would be suitable for fish migration.	163
Figure 6-36	Isometric plot from CFX-5, showing the behaviour of the thermal plume, in this case clinging to the foreshore along the left-hand-side of the plot. This type of plot can be used to explore the risk of creating a barrier to juvenile fish movement along the intertidal foreshore	163
Figure 6-37	Depiction of a diffuser outfall with ten discharge ports. Arrow indicates tidal flow (reversible).	164
Figure 6-38	CFX-5™ model outputs depicting dispersion of heat from a multiport outfall. (Upper: plan; lower, vertical section)	164
Figure 7-1	Example of sampling facilities for impingement and entrainment monitoring. Upper drawing shows diversion from FRR launder into water-retaining collection basket for impingement sampling. During sampling, shutters or gates are adjusted to divert flow into sampling well. Lower drawing shows the arrangement in vertical section, along with the suspended plankton net used for entrainment sampling.	178

1 Introduction

1.1 Purpose of study

The Health and Safety Executive (HSE) and Environment Agency[1] are working together to make sure that any nuclear power station built in the UK meets high standards of safety, security, environmental protection and waste management (Generic Design Assessment, GDA). There is a need to investigate cooling water options for new reactors and evaluate the environmental impact of these in terms of thermal, chemical and radionuclide pollution, and impact on biota. This document draws together information that will assist the regulatory agencies in this process.

1.2 Background

Regulation of the nuclear power industry in the United Kingdom is the joint responsibility of the Environment Agency and the Scottish Environmental Protection Agency (SEPA) (environmental) and the Health and Safety Executive (nuclear health and safety), that now includes the Nuclear Installations Inspectorate. The Government is committed to allowing the construction of new nuclear power stations provided they are subject to the normal planning process for major projects (under a new national planning statement) and provided also that they receive no public subsidy. The Government will complete the drafting of a national planning statement and put it before Parliament and, if approved, clear the way for planning applications for new nuclear power stations. This policy creates new challenges for the regulatory agencies given the variety of options available, since the development of nuclear power stations is now open to commercial competition. Hitherto all nuclear design and construction has been under effective government control, via the former Central Electricity Generating Board (CEGB) or, until 1996, the state-owned Nuclear Electric (NE) plc. Sizewell B was commissioned before NE was sold-off as British Energy.

A Strategic Siting Assessment of potential sites for nuclear new build has been carried out using exclusionary and discretionary criteria which were consulted upon publicly. This included a criterion on access to suitable sources of cooling. The Environment Agency advised the Government on whether sites were potentially suitable against this criterion. The public has been consulted upon whether sites are potentially suitable and the Government is currently considering responses to the consultation. Sites which are potentially suitable for deployment by 2025 will be listed in a Nuclear National Policy Statement. Should individual applications come forward for development consent, the Environment Agency will consider cooling as part of licensing with site specific detail.

Most people would be astonished at the demand for cooling water (CW) imposed by a nuclear plant, or indeed any other thermal power station. Laws of thermodynamics dictate that for every megawatt of power generated, up to two megawatts must be discarded as low-grade (low temperature) waste heat. In fossil-fuelled plants such as combined-cycle gas turbine (CCGT) stations and the new generation of supercritical coal plants significantly higher efficiencies (above 45 per cent) can be achieved, and in some cases low-grade heat can be put to use in district heating schemes, commercial horticulture or aquaculture. For nuclear stations, however, thermal efficiencies remain relatively low (around one-third efficient) and opportunities for constructive use of waste

[1] See Joint Regulators website: http://www.hse.gov.uk/newreactors (viewed 12/04/09)
[3] http://www.ospar.org/content/content.asp?menu=00220306000063_000000_000000

heat around isolated sites are fewer. Thus, a nominal 1,000 MWe nuclear generating station must discard nearly 2,000 MW into the environment as waste heat.
Heat disposal from thermal power stations can be into the atmosphere via cooling towers, via once-through direct cooling systems into surface water bodies and thence into the atmosphere or via hybrids that combine both methods. Direct cooling is the most efficient in terms of energy use, and therefore in terms also of greenhouse gas emissions. All UK nuclear stations built before today, and a large number of fossil-fuelled stations, discard their heat to water via direct cooling, the key reason favouring construction on the coast or estuaries.

Under the EU Integrated Pollution Prevention and Control 'IPPC' Directive (96/61/EC), the Environment Agency is obliged to consider, for designated installations, whether the technologies and techniques used by the developer would be Best Available Techniques or BAT. Technical guidance on BAT is given in BAT Reference or BREF notes. The European BREF document on industrial cooling systems (BREF Cooling; adopted December 2001) considers water cooling as the preferred option (as it helps reduce emissions of greenhouse gasses) and defines direct cooling as a BAT for large combustion plants in coastal locations, provided that the aquatic ecosystem is not adversely impacted. While nuclear power stations are not within the scope of the IPPC Directive, the BREF notes are useful to inform the Environment Agency's discharge consenting process. OSPAR (Oslo and Paris Conventions) also requires the use of BAT when considering prevention and elimination of marine pollution. OSPAR covers discharges of radioactive substances[3].

More recently, however, the validity of the BAT definition of direct cooling has come under challenge, owing to what some believe may be unacceptable environmental impacts on the source and receiving water bodies. Pressure on this issue has emanated largely from activities in the USA, where impacts arising from abstraction of cooling water are regulated under the Clean Water Act s.316(b). Legal actions brought against the US Environmental Protection Agency (EPA) by the pressure group Riverkeeper, Inc. have forced the EPA to act on this issue. In its 2004 decision in Riverkeeper v. EPA (Riverkeeper I), the Second Circuit upheld the EPA's Phase I regulations, which set closed-cycle cooling systems (wet or dry tower-cooling) as BTA (best technology available) for new power plant cooling facilities[4]. In this case, the finding relates to effects of the abstraction only, and not of the discharge of heated effluents into the environment. In the UK, the validity of direct cooling as BAT for a new 2,000 MWe power station proposal falling within a Natura 2000[5] site, was investigated on behalf of the Countryside Council for Wales (Cambrensis, 2008). A key conclusion from the Cambrensis report was that, since the latest information to the 2001 BREF Cooling note was published in 1997, it is now out of date given developing techniques in indirect cooling. However, the same could be said of environmental mitigation techniques against abstraction impacts, an aspect which is not considered in the Cambrensis report. To give proper consideration to this issue, a thorough understanding of the technical background and the types and levels of environmental impacts are required, along with an appreciation of recent developments in cooling technology and associated mitigation techniques.

Effects upon the aquatic environment from direct cooling water systems, that is those that do not use cooling towers, relate primarily to two causes:

- The incidental capture of organisms contained in the abstracted cooling water stream: these may be divided into two components, the first caused by impingement of fish and invertebrates on cooling water filter screens and

[4] *Riverkeeper I*, 358 F.3d 174, 194 (2d Cir. 2004).
[5] Milford Haven Special Area of Conservation

the second, entrainment of mainly planktonic stages of fish, invertebrates and microscopic plants which pass through the screens through the cooling circuit, before being discharged back to the wild via the thermal discharge.

- The effects of the thermal discharge, which sometimes contains residual oxidants and byproducts from use of biocides (mainly chlorine-based) in the receiving water bodies.

Where cooling towers are used, these impacts are diminished but arise, for example, from cooling tower plume emissions and visual impact and increased energy use (or decreased efficiency of the power plant). This report helps to explain why and how water is used for power station cooling, what the technical options are and the main design considerations, types and levels of environmental impact and mitigation techniques now available.

1.3 Terms of reference

The following objectives were established for this study:

1. to give a brief overview of all power station cooling water systems in use in the UK and abroad;
2. to identify and give details of all cooling water options for new nuclear power stations in the UK;
3. to give an overview of the generic environmental issues associated with cooling water systems of nuclear power stations (for example thermal, chemical and radionuclide pollution, fish and invertebrate intake);
4. to identify and explain any environmental issues associated with specific cooling water options;
5. to evaluate the cooling water options in terms of environmental concerns and assess the best options for different types of water body (coastal, estuarine and freshwater).

1.4 Sources of Information

The study uses only publicly available documents and references, all of which are listed in the text or in the reference list at the end of the report. Where internet sources have been used, the dates on which these were accessed are given. A number of internal CEGB and power industry reports have been cited, which should be available from the British Library but are also held by Jacobs Engineering, Southampton Office.

2 Why power stations need cooling

2.1 Thermodynamics and the steam cycle

Power stations are essentially factories that make electricity, the value of which exceeds the cost of its production. As with any manufacturing process, energy has to be used in making the product – but in thermal power stations two or three times more energy goes in, as fuel, as comes out as electricity. However, the versatility or quality of electrical energy is far more valuable in practical and monetary terms than the energy in the fuel. Few people could find much use for a lump of poor-quality coal or a bucket of residual fuel oil, let alone a chunk of uranium.

The benefits of electricity do not come without costs, the two most obvious and most heavily criticized being this low rate (efficiency) of conversion of fuel energy into electricity, and the associated discarding of unusable heat into waterways and the atmosphere. These issues - efficiency, water use and disposal of waste heat - cannot be understood without a brief look at the components and workings of a thermal power station and, particularly, some understanding of thermodynamics.

The First Law of Thermodynamics says that energy can be neither created nor destroyed but only converted from one form to another. Although the conversions might leave the total *quantity* of energy unchanged, the Second Law says that the *quality* of energy will decrease at each conversion and eventually becomes degraded to the point where it can no longer do useful work. For example, in hydroelectric generation there is more opportunity to obtain work from the potential energy in a small volume of water at 0.5 km altitude on a hillside than from a 1,000 times greater volume at 0.5 m above sea level. In this case it is the difference between the initial altitude and sea level that determines the utility of the energy. By analogy, the work that a steam turbine can extract from steam is determined by the difference between its inlet (heat addition) and exhaust (heat rejection) temperatures and, by this same analogy, there is more scope for extending the working temperature range *upwards* than *downwards*.

Thermal power stations use water as the *working fluid* in a four-stage vapour power cycle - the Rankine Cycle - during which it is alternately vaporised and condensed. Vaporisation (steam raising) needs heat. This comes predominantly from burning coal, oil or gas and to a much lesser extent from sewage and landfill methane, biomass, domestic refuse, solar and geothermal sources, and from the controlled splitting of uranium atoms in a nuclear (fission) reactor.

The power cycle is probably best visualised by reference to coal being burnt in a tube and drum boiler. The coal's chemical energy is transferred, via the thermal energy (radiance) of flames and hot combustion gases (conduction), to water circulating under pressure through vertical steel evaporator tubes lining the boiler walls. Because of the pressure, the temperature at which the water boils[6] is well above 100°C – for example, 450°C at 100 bar. Continued heating provides the latent heat of vaporisation (enthalpy

[6] Water molecules are in a constant state of agitation. At room temperature and pressure, some molecules briefly burst through the water surface creating a small *vapour pressure*. With increasing temperature the molecules become more energetic, more escape the surface and the vapour pressure rises. When vapour pressure equals ambient (air) pressure more molecules are leaving the surface than are re-entering and the water boils. Increasing (or decreasing) ambient pressure raises (or lowers) the boiling point.

of evaporation) to convert some of the water into bubbles of steam. The tops of the tubes are welded to the underside of a horizontal drum spanning the width of the boiler. As the steam/water mixture erupts into the half-full drum the steam separates out. The water is routed back to the *boiler feed pump* for another pass through the boiler, whilst the steam goes to the turbine.

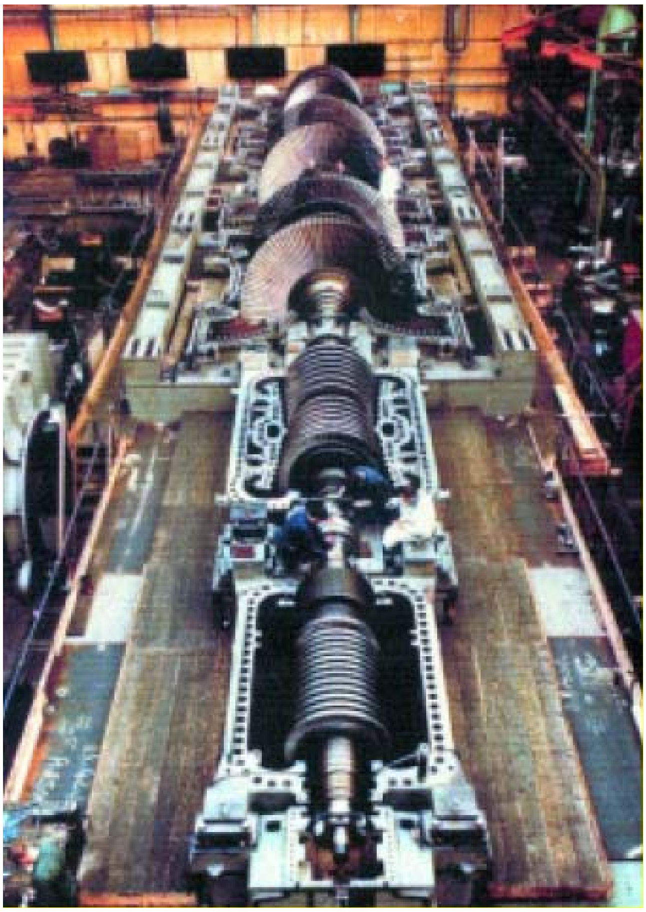

Figure 2-1 Internal view of a 500 MWe turbine. Small-diameter HP cylinder is in the foreground, then an intermediate pressure cylinder and three LP stages. Overall length of unit is 55 metres and large LP disc (blades) diameter is 2.5 metres.

Here, its temperature and pressure progressively decreases whilst its volume increases. During this expansion the steam loses thermal energy but gains velocity and kinetic energy that is transformed into work as it impinges against the blades of the turbine rotors[7]. These are coupled to a generator that produces electricity. At the far end of the turbine, the steam encounters a cold *condensing surface* where a volume of around 8.5 m^3 of steam rapidly contracts to a mere one litre of water, producing a substantial (40 mbar) vacuum[8] or "backpressure". Just as increasing the pressure raised the boiling point; so decreasing the pressure reduces the boiling point, or in this case the condensation or dew point, of the steam. At this level of vacuum, condensation occurs at around 30-35 °C. The cycle is completed when the boiler feed

[7] Most "turbines" actually comprise up to five separate turbines or "cylinders", often on the same shaft, operating at two or three pressure ranges – typically one high pressure HP, one intermediate IP and three at low pressure LP. The intermediate pressure cylinder is usually omitted with low quality steam.

[8] A pressure of 40 mbar absolute is -960 mbar gauge, but when quoted as backpressure the sign is generally ignored. "Gauge" is measured relative to normal atmospheric pressure so zero mbarg is approximately 1,000 mbara.

pump injects the condensate back into the base of the boiler, adding mechanical energy (pressure) to the system.

The cycle cannot continue unless the latent heat given up to the cold surface during condensation is removed continuously from the system. This is where the cooling water is needed. The volume of cooling water required is determined by the *weight* of steam to be condensed. Consequently, increasing the amount of work that can be got out of each kilogram of steam (efficiency) before it has to be condensed enables more electricity to be generated for a given volume of cooling water and, crucially, reducing the quantity of heat discharged to the environment. The condenser-cooling water does not come into contact with surfaces at temperatures above 35°C.

2.2 Improving efficiency

Improving conversion efficiency is probably the simplest way to minimise heat rejection. "Simplest" does not refer to the technological challenge, but to the political will to carry it through. Since conversion efficiency determines fuel consumption there is a strong financial incentive involved that, fortuitously, coincides with the IPCC BAT for cooling - to minimise the need for cooling by improving process efficiency. At 33 per cent thermal efficiency, 7.3 MJ of heat is rejected per kWh electrical output, falling to 6.4 MJ kWh at 36 per cent thermal efficiency. Strategically, the most thermodynamically efficient equipment need not necessarily be the best option; similarly, the balance between current and predicted interest and discount rates, between capital and lifetime running costs and between reliability and efficiency all can influence choice.

2.2.1 Theoretical efficiency

The maximum theoretical efficiency of most of today's large sub-critical steam turbines is around 60 per cent (Table 2-1) although in practice this is reduced by heat losses and friction. If the losses at every step, from boiler to generator, are included then the overall efficiency of conversion falls to around 40 per cent. This may look poor but is a great advance on the 11 per cent attained pre-1918 and 20-25 per cent in the 1950s. By way of comparison the last railway steam locomotives were about 12 per cent efficient and typical modern petrol and diesel engines are 20-30 per cent efficient.

Table 2-1, calculated using a standard rejection (exhaust) temperature, shows not only an increasing theoretical efficiency for turbines over time but also a convergence between this and the station's overall generation efficiency. These improvements have been possible through better understanding of theory and better design, coupled with new alloys and materials that tolerate higher temperatures and pressures.

Table 2-1 Improvements in theoretical efficiency and in performance, using a constant 35°C turbine exhaust temperature

Turbo-generator rating and approximate date		Steam inlet temperature		Carnot cycle efficiency (theoretical)	Typical overall generation efficiency	Ratio of overall to theoretical efficiency
MW	Date	°C	°K	%	%	%
11	~1914	345	618	50	*ca* 15	30
30	~1929	370	643	52.1	20	38
46	~1949	441	714	56.8	22.4	39
120	~1962	538	811	62	32.1	52
660	~1982	541	814	62.2	36.9	59
500	~2000	600	873	64.7	40	62
Supercritical		700	973	68.3	45	66

These developments have proceeded in parallel with the use of larger boilers, turbines and generators. In fact, increased size (ratings) and better steam conditions are complementary and mutually dependent; it would be as uneconomic to build a 30MWe unit to operate at 540°C/160 bar as building a 660°C MWe unit to operate at 350°C/40 bar. The driving factor behind this progression is economics – again best illustrated by coal-fired stations where fuel represents about 95 per cent of the production cost (Table 2-2 – after Tombs, 1978).

Table 2-2 Increasing production (kilowatt-hours) from one tonne of coal

Date	kWh generated per tonne of coal burnt
1900	*ca*. 200
Pre-1914	555
1920	641
Pre-1939	1,591
1977/78	2,088
2000 Drax	2,700
2001 Aberthaw B	2,654
2001 Didcot A	2,652

An efficiency gain of even a fraction of one per cent translates into significant savings, bringing benefits right across the environment, since not only is less cooling water abstracted and less heat rejected but also, in the above example, less coal has to be mined and transported; less ash, carbon dioxide, sulphur dioxide and nitrogen oxides are emitted. To date, fuel cost is less of an issue for nuclear stations.

2.2.2 Superheating, reheating and regenerative heating

The steam cycle described in Section 2.1 is basic and many modifications can improve efficiency. The boiler drum delivers saturated steam at the temperature and pressure of the water from which it was separated. Any slight decrease in temperature or pressure, as will inevitably occur en route to the turbine, will result in condensation and the formation of droplets that, for a variety of reasons, are undesirable. This can be avoided by first passing the steam through a moisture separator or, more commonly, a bank of superheater tubes in the hottest part of the boiler. Any carried-over droplets are evaporated and the steam temperature is raised, without change of pressure, far above its saturation temperature at this pressure. It now behaves as a gas and obeys the ideal gas laws relating temperature, pressure and volume. Superheating also adds more thermal energy to the steam, thereby further increasing its capacity for work in the turbine where its expansion, accompanied by the release of this superheat energy, proceeds until it reaches a saturated condition. Steam leaving the first (high-pressure) cylinder of the turbine is often passed through reheater tubes in the boiler before entering the intermediate pressure cylinder, following which it may undergo a second reheat. For a given energy input, the output (efficiency) can be increased by selectively boosting temperatures along the water-steam cycle. Steam is also bled off at various positions between the boiler and condenser for a range of ancillary heating purposes and, when practically all of its sensible heat has gone, latent heat is recovered by injecting it into the cold condensate returning from the condenser to the boiler. This regenerative preheating not only improves steam-cycle efficiency but, by condensing a significant weight of steam directly into the condensate, also reduces heat rejection via the cooling water. In a typical 500ºMWe conventional unit (Fawley) 32 per cent of the steam applied to the turbine does not enter the condenser, whilst Leizerovich (2005) suggests that 45 per cent of the steam flow from a pressurised water reactor (PWR) is used in reheaters and regenerative heating.

2.2.3 Efficiency and size

Since the early days of the industrial revolution, it has been known that the cost of a machine is roughly proportional to its weight, and that weight and size do not increase pro rata with output. This means that the pumps, boilers, turbines and generators require relatively smaller sites, foundations and buildings. Apart from reducing capital costs, increasing the size of a machine can increase its efficiency by reducing heat losses and friction. The first (1890s) power stations cost, at current prices, about £9,000 per kW of installed capacity but by 1965 the cost, of vastly more complex and efficient equipment was around £900 per kW.

The progression to larger units, with higher temperatures and pressures and lower cooling water requirement, has not proceeded uninterrupted. In 1947, to speed postwar reconstruction, UK procurement was standardised around 30 and 60 MWe units (350°C/40 bar), increasing to around 120 MWe (530°C/100 bar) and to 350 MWe by the end of the 1950s. From the mid-1960s, 500 and 660 MWe units (540°C/160 bar) were specified for new conventional stations. However Magnox (GC1 nuclear) stations, commissioned between 1955 and 1973, had low design steam conditions (350°C/48 bar at Bradwell) because their fuel elements were limited to under 450°C. This required turbines similar to those installed from the 1920s to 1940s. AGR (GC2 nuclear) stations have gas temperatures around 640°C and design steam conditions and cooling water requirements comparable with contemporary conventional plants, using similar 660 MWe units. Sizewell B PWR, commissioned in 1995, represented another reversal of the higher pressure/higher temperature trend, using a pair of modified 660 MWe units operating at 282°C/66 bar. In the UK there are no units larger than 660 MWe, although there are many overseas, and no units operating supercritically above 221 bar - the so-called "critical pressure" for water (see Glossary).

2.2.4 Supercritical operation

The boiling point (saturation temperature) of water increases with pressure up to 221 bar - the critical pressure. At this pressure, the boiling point is 374°C but this remains unchanged by any further increase in pressure. Thus 374°C is the highest temperature at which liquid water can exist since, at and above the critical pressure, no latent heat (enthalpy of evaporation) is needed to convert liquid water to vapour [contrast this with the 2,258 kJ kg^{-1} required at atmospheric pressure]. "Supercritical" is a thermodynamic term describing this state where there is no clear distinction between liquid and gaseous phases.

One advantage of supercritical operation is that the boiler tubes yield a single phase fluid that can pass directly to the superheater, enabling once-through steam generators[9] to be used. There is no need for a heavy, thick-walled drum to separate water from the steam.

No current design of nuclear power station operates supercritically[10] and lightwater reactors (PWRs and BWRs) in particular operate far below it (under 70 bar). However the Environment Agency will be assessing proposed supercritical coal-fired plant, such as the 2 x 800 MWe Kingsnorth B and Tilbury B stations, and comparisons will be drawn with the relative volumes of cooling water required by nuclear stations. Although no existing UK conventional stations operate supercritically, 375 MWe supercritical units were operating at Drakelow C (UK) in 1960, with steam conditions (595°C/250 bar) comparable with the most advanced units available today. However, lacking the benefit of today's materials, they had poor reliability whereas current state-of-the-art supercritical coal-fired plants have a reliability comparable with subcritical plant and efficiencies above 45 per cent. Indeed, ultra-supercritical plants, operating at 700°C/350 bar with 720°C reheat and approaching 50 per cent efficiency are now considered feasible (DTI, 1999).

2.2.5 Nuclear thermal cycles

In PWRs lightwater[11] serves as the moderator and *primary coolant*. The fuel elements heat it to about 320°C but it does not boil since it is held under pressure (150-170 bar). It passes from the reactor to the base of a vertical, cylindrical steam generator and circulates through inverted U-tubes in the middle (evaporator) section where it transfers heat to feedwater in the *secondary coolant* circuit before returning to the reactor at about 290°C. The pressure on the secondary side of the U-tubes is around 69 bar, so the feedwater boils. This steam/water mixture enters the upper section of the steam generator, passing first through a steam separator (moisture separator) that imparts a spin to remove the larger entrained water droplets and then through steam driers. The driers comprise a series of angled louvres (chevrons) in which the steam flow is forced to make several sharp changes in direction, throwing droplets into contact with their surfaces. The dry steam (about 0.25 per cent water content or 99.75 per cent steam quality) enters the HP turbine in a nearly dry saturated condition but becomes wet as it expands. At Sizewell PWR (282°C/67 bar) additional moisture separators are sited

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[9] "Boiler" seems inappropriate since no boiling occurs.
[10] The Canadian **CANDU X** will operate supercritically but will not be in commercial operation before 2020.
[11] Ordinary H_2O, as distinct from heavy water D_2O.

between the high-pressure (HP) and low-pressure (LP) cylinders, together with reheaters, heated by steam bled directly from the steam generator. This avoids excessive wetness that would erode the LP blades[12] enabling LP cylinders similar to those of standard 660 MWe sets to be used. The Wolf Creek and Callaway (USA) four-loop SNUPPS[13] stations, on which the Sizewell design was based, use a single 1,300 MWe turbogenerator operating at half-speed (1,800 rpm for 60 hz) to reduce tip speed and erosion of the LP blades.

In boiling water reactors (BWRs) lightwater is the moderator and only coolant. The internal layout of the reactor vessel resembles the steam generator of a PWR in that it is only part-filled by water. The submerged fuel elements heat the water to some 290°C that, since the reactor is at a pressure of about 70 bar, causes it to boil. The water-steam mixture passes through separators and driers located above the core, before being fed into the turbine. Steam conditions for PWRs and BWRs are physically very similar but the latter can carry contamination into the turbine and condenser.

With AECL CANDU[14] out of contention, UK regulators are considering a PWR design by Areva-EDF; another PWR by Westinghouse and a BWR design by General Electric-Hitachi.

2.3 The role of the condenser

It was noted, in section 2.1, that the condensation of exhaust steam creates a vacuum (backpressure) that increases generation efficiency. Hence the cooling system must be capable of *continuously* and *consistently* rejecting the heat load necessary to maintain the condensation temperature corresponding to the optimum turbine backpressure. For a given steam flow and cooling water flow, the basic design (and operating) parameter for the condenser is the temperature of the incoming cooling water, which is variable. However the condenser must remain capable of transferring the necessary heat load even in summer. *Undercooling* - operating above the design point - decreases efficiency and leads to frictional heating and expansion of the last-stage turbine blades (this can result in loss of clearances) as they rotate through an over-dense atmosphere of steam. *Overcooling* - operating below the design point - might increase efficiency but at the risk of extending the condensation zone into the turbine, causing droplet erosion of the blades. Moreover the intention is to condense steam, not to cool the condensate as it drips down through the tube bank: any heat lost here will have to be replaced by the boiler. In any case, once the velocity of the steam through the turbine exhaust passages reaches sonic velocity (choking flow) there is no further backpressure gain to be made. Inevitably such considerations result in a series of trade-offs in turbine and condenser design reflecting capital and running costs across a range of conditions.

Condenser performance is also reduced by factors such as scale on the outside (steamside) of the tubes and by slime and scale (waterside fouling) inside. These effectively insulate the tube wall. A major backpressure problem for some condensers is steamside air ingress. Air can outgas from the boiler feed water during intermittent operation and also be pulled in around the LP turbine exhausts and past flange seals on the waterbox doors. Condensers need vacuum pumps to remove air from the

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[12] Since there is less energy in the steam, LP blades must be longer than HP blades and, at 3,000 rpm (50hz) the tips may be travelling at 350 ms^{-1} and are prone to erosion and unbalancing by impact with water droplets.

[13] Standard Nuclear Unit Power Plant System. A Westinghouse design having four steam generators (four-loop).

[14] AECL Atomic Energy of Canada Limited; a second-generation heavywater cooled and moderated reactor. Withdrawn from UK selection process in March 2008.

turbine cylinders and condenser when the unit is started up, but it is not uncommon for them to have to be run continuously. Air is non-condensable and accumulates at the top of the condenser, preventing steam from reaching the top rows of tubes (air blanketing). When the system is unable to reject the heat load necessary to achieve a steam condensate temperature corresponding to the optimum design turbine backpressure there are two penalties:

- Efficiency loss – for a given fuel input, the electrical output is lower. An increased heat rate (fuel burn) would be needed to restore output.
- Capability loss - the unit cannot handle the design heat input (steam flow) within the backpressure limit .The steam flow must be restricted to restore the backpressure, so the generator does not reach its rated value (loss of output).

2.3.1 Can a thermal power station work without steam?

In theory, yes. Water is not the only working fluid although it has many advantages. Alternative fluids (such as ammonia and a variety of organic compounds) all are more expensive and have one or more of the hazards conspicuously absent from water. However, an organic Rankine cycle (Karellas & Schuster, 2008) can use low temperature heat sources and be efficient in small-scale applications. Most proposals envisage using these fluids to boost parts of the Rankine steam cycle, rather than wholesale substitution for water. The highest primary coolant temperature in existing large nuclear stations is 640-650°C (gas in UK AGRs) but prototype high temperature gas reactors (HTRs) using ceramic fuel indicate that far higher temperatures are feasible and could be used to run gas turbines. The gas would expand and cool in the turbine and the exhaust would be recycled to the reactor.

2.3.2 Is water essential for cooling?

No, but it is extremely effective and very convenient. The ultimate heat sink is the atmosphere (moist air) so the choice of coolant is water or air. Water has a high *specific thermal capacity* and is able to absorb large amounts of heat with little increase in temperature, whereas the thermal capacity of air is a quarter that of water (1.0035 J kg^{-1} against 4.186 J kg^{-1}). Moreover its density is 830 times less. It is seldom economic to build coolers (essentially car radiators on a grand scale) of sufficient size for the recool temperature to closely approach air (dry bulb) temperature. Consequently air-cooling is the most efficient and least-cost option only when air temperature is very low for much of the year or when the temperature of the fluid to be cooled is above 55°C. Even so, dry cooling is used for power stations, even in areas where adequate water is available (section 2.4).

2.4 Principles of cooling

Heat transfer depends on the existence of a temperature gradient. In the condensers discussed above, steam at about 30°C impinges upon cold (say 5°C to 20°C) water-filled tubes where it condenses, dumping its latent heat in the film of water on the tube surface. This heat is conducted across the tube wall and into the cooling water. The following options are discussed in more detail in the next chapter (3).

2.4.1 Direct (once-through) wet cooling systems

In a direct system the cooling water is discharged, dispersed and diluted in a river, lake or sea. Ideally none of the discharge finds its way back to the intake, or at least not until its temperature has been reduced to near-ambient. The ultimate heat sink is the atmosphere, via convective and evaporative transfer from the water surface - a process that is enhanced by wind and waves. This is the least complex, generally most thermally efficient and often cheapest option, with the added advantage of being visually unobtrusive. In some cases a lake is "engineered" by means of bunds to create a longer path between outfall and intake – indeed the lake may have been created for just this purpose – although it may acquire secondary functions such a recreation and water storage. Cooling canals are a linear lake, with the water following a tortuous path from the outfall to the intake. The cooling capacity of lakes and canals can be increased by agitation of their surface and by fountains (sprays) that increase the effective surface area for sensible (contact) cooling and for evaporative cooling (Section 3.1.4).

2.4.2 Indirect (recirculating) wet cooling systems

With indirect cooling, the transfer of heat to the atmosphere is accomplished within the cooling circuit. Most large recirculating circuits use *wet towers*. These improve upon the sprays described above by creating a directional air flow and by maximising air contact with the falling water droplets. This enables the cooling pond to be reduced to no more than the basal area of the tower, with a significant decrease in the volume of water held in the circuit. A wet tower is not only cheaper to construct than a dry system but it also provides lower recool temperatures and needs less maintenance. The degree of cooling depends upon the wet bulb temperature of the air (T_{wb}). A low T_{wb} indicates cool air, low humidity or some combination of these; the lower the T_{wb} the greater the scope for evaporation and evaporative cooling. Towers can function even when the inlet air is at its wet bulb temperature since the air entering the tower is warmed (sensible heat) by contact with the water, thereby raising its temperature so that it is no longer saturated. The minimum recool temperature would be the wet bulb temperature of the air entering the tower (100 per cent wet bulb performance). In practice this can never be realised and even at low wet bulb temperatures a 65-70 per cent performance is considered good.

At the design air:water loadings, the *cooling range* of a tower (cooling water inlet temperature minus the outlet or recool temperature) is relatively immune to changes in wet bulb temperature. The difference between the wet bulb temperature and the recool temperature is described as the *cooling approach* or just *approach*. A high air:water ratio needs a large tower and large airflow, but permits a close approach to wet bulb temperature.

2.4.3 Natural draught towers

In a *natural draught tower* the tower shell extends high above the pack (wet section) and functions as a chimney, with the convective rise of warmed air drawing in cooler air at the tower base. With most natural draught towers, there is little if any scope for air control and this often results in overcooling. Conversely, should the air temperature (dry bulb) exceed the temperature of the inlet water, the air flow will stop. A *hybrid tower* [15] - a natural draught tower that can be assisted by fans at the base - has a greater cooling capacity than a natural draught tower of similar size.

[15] Various other systems are also now described as "hybrid".

2.4.4 Mechanical draught towers

Mechanical draught towers rely on fans to push or pull air through them, which gives a more consistent air flow than in natural draught towers though the fans increase the running costs. By using internal doors and/or variable speed or variable pitch fans, the performance of such towers can be matched to prevailing atmospheric conditions. One disadvantage of the low profile of mechanical draught towers is that any plume of vapour or drizzle emerging from the top of the tower is closer to the ground than with a natural draught tower. Where this is seen to be a problem, a dry section (radiator) can be built above the wet section to raise the saturation temperature of the plume. In some climates, the tower can operate for much of the year using the dry section alone.

2.4.5 Dry coolers and condensers

As the name suggests, the water does not contact the air – as in a car radiator. Heat is transferred to air by conduction as *sensible heat* or “heat that can be felt”. Since heat transfer depends upon the existence of a temperature gradient, the recool temperature will always exceed air temperature (dry bulb temperature - as given in weather forecasts). With an infinitely large radiator and unlimited air to pass through it, it would be possible for air and recool temperatures to approach to within a fraction of a degree. In practice, a balance must be struck between the benefits of a lower recool temperature and the size/cost of the radiator. Air cooling can be used for condensing, not just for cooling. This was originally used for relatively small-scale applications but increasingly is being used for multi-megawatt power plant. As a rule, air cooling is the most efficient and lowest cost option where the input temperature is above 55°C and the year-round air temperature is low. It is possible to lower the recool temperature and to reduce the size of an air/water cooler by spraying water over it (WSAC - wet surface air cooling). As the water evaporates, it removes *latent heat* from the cooler surface.

2.5 Economics

For many years, the main reason for choosing tower cooling for a power station was that insufficient water was available or readily accessible for once-through cooling. In some cases this was seasonal and towers were brought into use only in summer. The evaporation of one kg of water removes sufficient heat to cool 100 kg of water through 10°C; thus at minimum a tower cooled station need abstract only one per cent of its circulating volume. In practice about three per cent is usually abstracted, of which one per cent is evaporative loss and the other two per cent is discharged back to the waterway to prevent build-up of dissolved and suspended solids in the circuit. However, a whole raft of considerations must be taken into account, some of which might appear retrograde in terms of technology or efficiency. Tower cooling permits more flexible siting, thereby reducing fuel transport and/or transmission costs, and increasingly is seen as a way of avoiding potential delays and litigation sometimes associated with licence applications for direct cooling. Against this are the higher capital costs for towers although these costs might be marginal when set against emplacing or tunnelling large diameter pipes for direct cooling. Similar, and often more severe penalties, attach to the use of air coolers and condensers. Running costs, particularly for mechanical draught towers and air coolers, are higher than for direct cooling and there is less thermal efficiency (higher backpressure) and, for conventional plant, greater carbon emissions because of higher cooling water temperatures at the condenser inlet.

However, the turbine LP section can then be cheaper, with fewer, smaller diameter discs and a corresponding reduction in the dimensions of the casing and exhaust steam passages. These economic arguments need to be weighed in each case.

3 Existing power station cooling systems

3.1 Critical review and description of alternative cooling circuits

3.1.1 Summary of large (above 1,000 MWe) UK cooling circuits

Appendix A provides a list of large UK power stations, along with details of their cooling systems. Reference is made to UK stations throughout the following text.

3.1.2 Cooling requirement

The previous chapter outlined how steam-electric power stations make use of the Rankine cycle, of which condensation is an essential stage. During condensation the enthalpy of evaporation (latent heat) of the steam is released and the cycle cannot continue unless this heat is removed from the system, which is most commonly and efficiently done by cold water flowing through tubes in surface condensers. Higher temperature heat sources enable a greater percentage of the heat to be converted to electricity, leaving less waste heat to be discharged. Similarly, lower temperature cooling water will also yield higher efficiency so a power station on the UK North Sea coast is more efficient than an identical one on the Gulf Stream-warmed south and west coasts. The volume of cooling water that is required is determined by the *weight* of steam to be condensed. Improved technology and metallurgy has steadily increased the amount of work (units of electricity) that can be generated from each kilogram of steam. This progression was driven by economics: primarily the cost and availability of fossil fuels. Fuel costs are less important for nuclear stations and, for technical and economic reasons, heat rejection (per unit of electricity) by lightwater reactors is high.

The principles involved in the layout and hydraulic performance of cooling circuits are considered in Chapters 4 and 5. Basically, the requirement is to deliver an uninterrupted, adequate supply of cold water to the heat exchangers and then to remove it and its reject heat. About 90 per cent of the water passes through the main cooling water system (MCW) to the condensers, with the balance entering supplementary or auxiliary cooling circuits. The layout, duties and names of these circuits vary from station to station. Increasingly stringent design at nuclear sites has necessitated the provision of discrete essential (reactor) cooling water systems (ECW or RCW). Most of these systems incorporate additional finer-mesh pressure strainers to protect small-bore coolers or plate heat exchangers (PHEs).

This chapter follows the order set out in *Appendix B* and is arranged more or less in order of decreasing water demand. The first section (3.1.3) considers the CW intake arrangements, mainly with reference to UK stations, and is broken down according to water source, whereas subsequent sections ignore the source and focus on the equipment.

3.1.3 Direct cooling

Intake position

This section gives a brief overview: engineering design aspects are detailed in the next chapter. Intakes in the sea or an estuary may be sited onshore, typically set flush into a quayside or seawall (Heysham A & B, Kingsnorth, Torness and Hartlepool) or at the head of a short canal (Fawley, Dounreay PFR). Most onshore intakes have a dredged channel leading to them that may or may not be shared with ships. Nearshore intakes are often at the end of a short jetty, with shafts and tunnels connecting them to the onshore CW pump forebays (Hunterston); Wylfa intakes are close to a jetty, but are not part of its structure. Offshore intakes have long tunnels from land, terminating either at a massive intake structure (Aberthaw, Hinkley Point A & B) or at downshafts (plug holes) in the seabed (Dungeness A & B and Sizewell A & B). These latter may be marked by a buoy or be surmounted by a platform (Sizewell A and, originally, Dungeness A). There is no hard and fast distinction, such as distance, depth or design between an offshore and a nearshore intake, but both types are surrounded on all sides by water.

Siting an offshore intake also has to take account of local factors such as proximity to shipping channels or fishing grounds and, if at the seabed, the need to mark it by buoy or a permanently lit topmark. It must not endanger swimmers, divers, sailboarders and small boats. Over the years, offshore intake design has been scaled down in size and complexity, partly in response to changing views on the need to isolate and dewater the shafts and tunnels for inspection and maintenance, but also to developments in diving and submersible vehicles that obviate the need to dewater.

Offshore seabed intakes

These usually have a vertical shaft that extends a metre or so above the seabed, so as to avoid drawing in mobile sand and gravel, with a cage-type screen on top to keep out trash (“parrot cage and plughole”). “Plughole” refers to the predominantly downward direction of the incurrent. These days it is more usual for the top edge of the shaft to have a horizontal lip extending to about twice the shaft diameter, with a solid “velocity cap” of the same diameter set about one metre above it, so that the incurrent becomes horizontal. The circumferential gap is protected by vertical bars to exclude trash. The bars may be welded into panels and fixed to the pillars supporting the cap or be set individually into the concrete at top and bottom. At Wylfa, the bars extend through the cap and can be withdrawn for maintenance. The Dungeness A and Sizewell A intakes were surmounted by platforms resembling small oil rigs on which stood a crane for removing the protective cages and lowering the shaft plugs stored on deck. The Wylfa intakes can be serviced from the nearby jetty; the shaft plugs remain underwater where they form part of the intake caps. Sizewell B intake is fully submerged and has no provision for isolation and dewatering. This decision was based on findings that during the previous thirty years few stations had needed to dewater their tunnels. Although Wylfa dewatered their tunnels “regularly” during statutory outages, Bradwell had done so “once or twice” but Sizewell A and Dungeness A had never done so. In fact the Dungeness platform was removed in the late 1980s, having become structurally unsafe - its crane had been condemned many years earlier.

Offshore headworks

Headworks such as Hinkley and Aberthaw are massive concrete structures, each designed to service more than one station (three at Hinkley) and can be reached on foot by a service tunnel. Both are surmounted by a crane for handling the removable grilles (guard screens) and stop gates. The central cylindrical concrete caisson of both structures was built on the nearby beach, floated out and sunk in position, after which the intake tunnel bores were completed and mated with the tunnel stubs on the caisson. The Hinkley structure was excluded from the previous (1980s) proposed C-station design since it would have been approaching 100 years old by the time that station was decommissioned. At Bradwell and Berkeley, the interlocking sheet piling of the construction caissons for the four intake (and outfall) shafts are incorporated into the final structure. Additional piling extends upstream and downstream as a “barrier wall” to prevent prompt recirculation of warm effluent. Both structures were equipped with gantry cranes for handling bar screens and the plugs (at Bradwell) and stop gates (at Berkeley) were stored on deck. The Berkeley tunnel access proved invaluable when the station was required to deploy 25 mm mesh smolt screens that required daily raising and cleaning over a six-week period every year.

Inshore headworks

The inshore (lower Thames) intake of Littlebrook D is a cylindrical sheet-pile tower that served as the caisson during excavation of the downshaft and tunnel. It is 20 metres in diameter with intake apertures around the entire circumference, covered by liftable 100 mm pitch bar screens. Grain (Medway) has a large rectangular concrete structure with two intake downshafts set in its base slab, 1.5 m above which is a solid “lower deck”. Around the periphery are liftable 250 mm pitch bar screens. The lower deck reduces the risk of vortex formation and the drawdown of floating oil and trash at low water. The two Hunterston stations and Longannet and Tilbury C have their intakes under a jetty, to the piles of which are fixed the guides for the liftable bar screens. Longannet has a fixed grille of 75 mm rods at 250 mm pitch to intercept and deflect heavier trash from the liftable 75 mm pitch bar screens immediately behind.

Onshore intakes

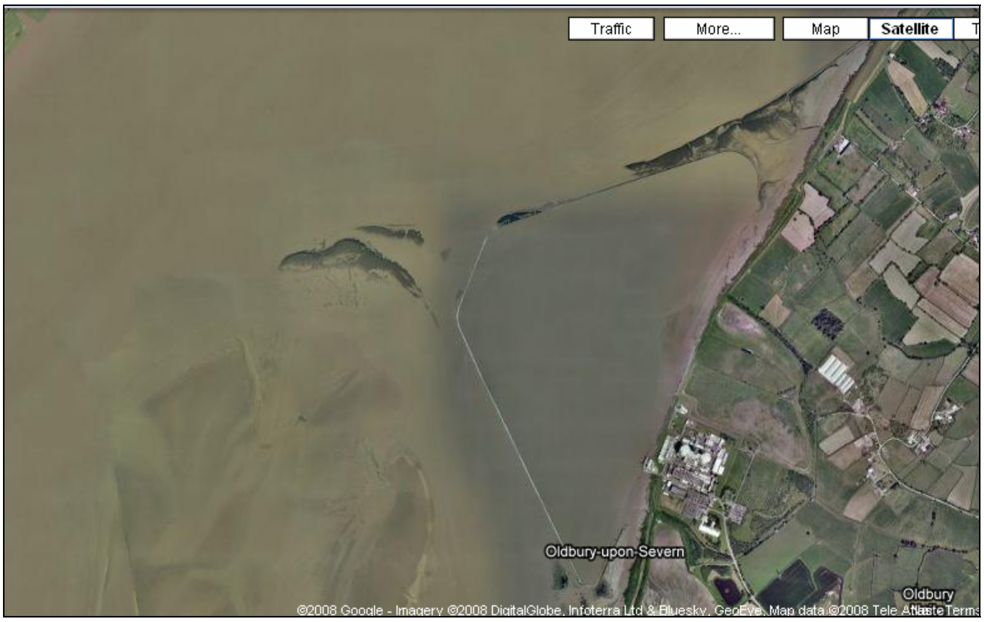
At many stations, these are “dead ends” where the cooling water is drawn into a confined channel towards a row of screens set across its path. Dounreay PFR and Fawley are probably the most extreme examples of this layout. At Heysham the intakes for the two stations are set at right angles to one another in the corner of a sheltered commercial harbour. There is a similar arrangement at Peterhead, although on a far smaller scale. Torness and Pembroke intakes are set in sheltered embayments, out of the main tidal flow and at Hartlepool the intakes are recessed deeply into the quayside. At none of these stations is there any possibility of trash being carried <u>away</u> from the screens by the tide. Only at Cockenzie and Kingsnorth, where the intakes are set flush with the dockside or seawall, is there any possibility of a flow across the face of the screens, and then only at high water when the surrounding mudflats are submerged. At both of these stations the incoming water is confined to a dredged channel at low tide. At Fawley there is an additional input of trash on a falling tide since the dredged channel intercepts many lateral creeks draining the surrounding saltmarshes.

Tidal reservoirs

The only UK example of a tidal reservoir is at Oldbury-on-Severn (

Figure 3-1) and no overseas examples were found.

Figure 3.1. Oldbury tidal reservoir: impounding wall is above level of falling tide



The mean spring tidal range in this part of the Severn is 11.3 m and the extreme range is 13.9 m (Extreme Low Water (ELW) is -4.7 m below Ordnance datum (OD). Given that the banks and channels in this reach are unstable, there was uncertainty over where to site the intake. An 8 km impounding wall (top +1.2 m OD) was built out on the mudflats to create a 200 hectare reservoir from which the cooling water is drawn. The design datum for the station's hydraulic gradient, and the minimum immersed depth for the drumscreens and CW pumps, is zero metres OD, so the extreme tidal range is limited to 9.1 m or less if the reservoir is not fully drawn down. When the tide falls below the top of the wall, approximately 2.4 million cubic metres are retained above datum (the "empty" level for operational purposes) that is sufficient for about 23 hours running on all four CW pumps or about 10 hours had the second station been built.

Water is abstracted through a wharf-type intake near the SE corner of the reservoir and discharged, via buried culverts, to outfalls outside the NW corner. Water can be recycled back into the reservoir by means of a variable crest weir on the southern outfall. When the tide rises above +1.8 m OD virtually all of the water from this outfall is recycled but since the wall has by then been overtopped by 0.6 m, the effluent is swept out of the impoundment. A recurrent problem has been that the reservoir acts as a settling pond. This was of little consequence when the accretion occurred in the "dead space" below the zero metre OD level but deposition was uneven and mud banks formed that threatened to cut off intakes. The solution was to use a pontoon-mounted pump to re-suspend sediment during the run of the tide so that it would be carried out of the reservoir, augmenting the natural tidal resuspension and deposition cycle.

River intakes

There are no UK rivers capable of cooling large (above 1,000 MWe) direct-cooled stations in their freshwater reaches. Even some tower-cooled stations (such as Willington, Notts, 800 MWe) found that during periods of low flow, a stretch of the Trent could flow upstream from outfall to intake. Since power station cooling is essentially non-consumptive the key issues tend to be environmental, such as effects on fisheries, rather than about water usage. In the USA power stations and agriculture each account for some 40 per cent of freshwater abstraction, although this becomes three and 81 per cent if consumption, rather than withdrawal, is considered (Feeley *et al.* US DoE, undated). In parts of the USA and continental Europe, ice must be added to the list of potential hazards at intakes.

Lakes/reservoirs

This section includes impoundments and natural lakes that are used for direct cooling. In some cases, the heat loading is insufficient to raise concerns about substantial ecological change but others are, in the words of the 1976 US Federal Register, simply recirculating cooling water bodies to which heat can be discharged without limitation. These latter would come into the category of "off-stream" cooling, along with spray ponds and cooling towers, but no examples have been found.

The design of cooling lakes and ponds seems to have been empirical and based on rule-of-thumb estimates such as one megawatt per acre (227 W m^{-2}) or on an average rate of heat loss of 70 Btu16 per square foot per hour (220 W m^{-2}). Kent (1938) reckoned that the heat dissipated from a still pond averaged 3.5 Btu $h^{-1}ft^2$ per °F temperature difference between the water surface and air. This would rise to 35 Btu $h^{-1}ft^2$ given a temperature differential of 10 °F (5.5 °C) and would double with a windspeed above 2.2 ms^{-1} (five mph). Chen *et al.* (1988) give a useful outline of the basic principles whilst Adams *et al*. (1990) deal with more complex issues. Lakes lose heat to the atmosphere by convection and evaporation, processes that are affected by factors such as flow rate, initial dilution by horizontal and vertical diffusion, vertical stratification, depth, length/width ratio, local topography, humidity and wind speed - much as for any other direct cooling system. The heat-loss performance at Trawsfynydd, the first and only lake-cooled power station in the UK, was extensively modelled, retrospectively, under various climatic conditions (see MacMillan, 1973 and Shepherd, 1973). This lake had been created in the 1920s for the Maentwrog hydro station but was modified in the 1960s with bunds to create a longer path between the outfall and intake of the 500 MWe Atomfa Trawsfynydd (closed in 1991).

[16] One Btu = 0.293 W = 0.252 kcal. One hectare (ha) = 2.47 acres (a)

Several small coastal stations, including Brighton A (Shoreham), Tir John (Swansea) and Roosecote (Barrow) used to be cooled from enclosed (non-tidal) saline docks, as is the replacement Roosecote CCGT (40 MWe steam). All these sites have acquired exotic flora and fauna and Natural England has recognised the importance of the Roosecote warm effluent to Cavendish Dock's ecology (Centrica, 2010), although it has acquired no special status.

Table 3-1 Cooling capacity of some reservoirs. For purposes of comparison nuclear stations are assumed to be 33%, and conventional stations 40%, thermally efficient.

Power Station	Installed capacity (MWe) and assumed efficiency [%]	Estimated heat rejection (MWht)	Lake area (acres)	Lake area (ha)	Capacity Ratio (MWe ha^{-1}) [MWe $acre^{-1}$]	Cooling duty (MWht ha^{-1})	Equivalent rate of net heat loss ($Wm^{-2}h^{-1}$)
Lake Anna USA	(2 units) 1,786 [33%]	3,626	9,600 including 3,400 acre hot lagoons	5,260 ha including 1,400 ha hot lagoons	0.34 [0.52]	0.69	69
Lake Anna USA	(4 units) 3,784 [33%]	7,683			0.72 [0.72]	1.46	146
Dresden USA	1,600 [33%]	3,248	1,275	516	3.10 [1.25]	6.3	630
Powerton USA	1,670 [40%]	2,505	1,442	584	2.86 [1.16]	3.78	378
Collins USA	2,520 [40%]	3,780	2,009	813	3.10 [1.25]	4.65	465
La Salle USA	2,156 [33%]	4,377	2,058	833	2.59 [1.05]	5.25	525
Braidwood USA	2,200 [33%]	4,467	2,539	1,028	2.14 [0.87]	4.35	435
Merom Lake USA	980 [40%]	1,470	1,550	627	1.56 [0.63]	2.34	234
Trawsfynydd UK	500 [33%]	1,015	1,090	442	1.13 [0.46]	2.3	230
European Nuclear Society (ENS)	1,300 [33%]	2,639	2,470	1,000	1.3 [0.53]	2.64	264
Range					0.72-3.10 [0.46-1.25]	1.46-6.30	146-630

Table 3-1 uses data from various sources. Heat rejection was estimated using a fairly generous 33 per cent thermal efficiency for nuclear stations and 40 per cent for conventional ones, but the former could be as low as 25 per cent. Consequently the heat inputs to these lakes may be conservative. The final column shows the hourly rate of heat loss necessary to maintain thermal equilibrium, over and above solar gain. The ENS example is more specific in that it seeks to maintain a 21°C recool (intake) temperature with 57 per cent relative humidity (wet and dry bulb, 8°C and 12°C respectively).

While evaporative water consumption is not worth considering at coastal sites owing to the vast volume of seawater available, losses from enclosed water bodies such as lakes can be more significant. A 1,000 MWe PWR station with a 2,000 MWth heat rejection could result in the evaporation of 400 kgs^{-1} (based on 'BREF Cooling' estimate of 20 kgs^{-1} per 100 MWth) or 1,400 m^3 per day, equivalent to 1 mm loss from a sea surface of 1.4 km^2. Such estimates are highly dependent on climate, weather and the rate of plume spreading. At North Anna lake (Table 3.1) the proposed Reactor 3 is to have wet tower cooling in summer to avoid overheating the lake – although the evaporative loss (from the tower) will be 66 per cent greater than for lake cooling. Reactor 4 will be dry cooled since it is considered that the evaporative drawdown of the lake at times of low rainfall would be excessive.

Creating a lake specifically to cool a power station does not avoid regulatory problems. In 1972 the North Anna River (Virginia, USA) was dammed to form a 5,260 ha reservoir to cool the (present) twin reactor 1,786 MWe North Anna station. The station abstracts 150 m^3s^{-1} and discharges it, some 9°C warmer, into three lagoons, totalling 1,400 ha, from which it finds its way back to the main lake. The entire lake is classified by the USEPA as a Class III water, in which the temperature must not exceed 32°C, but the lagoons are exempt since they are classified as a *waste (heat) treatment facility* (WHTF) by Virginia Department of Environmental Quality. As such they are not "a water of the United States". The lake, including the "hot side", is used for recreational boating and fishing - that is described as excellent - and there has been substantial residential development along the "cold side". There are now calls for the WTF exemption to be rescinded, arguing that (a) heat is recognized as a pollutant under federal law and (b) the US Congress has ruled that surface waters must no longer serve as waste treatment or waste conveyance systems. Once EPA has issued a criterion for water quality, individual states must adopt a corresponding criterion providing the same level of protection. A state can take into account only scientific considerations when determining water quality criteria, not economic and social impacts. If these appeals are upheld they would result in additional costs and/or energy penalties for this plant, stop the second one being built and probably affect all other lake-cooled stations – in the USA and, in time, elsewhere.

Cooling canals

This is the most "engineered" type of direct cooling system. Probably the biggest canal system is that at Florida Power & Light's Turkey Point facility. This was excavated to replace sea outfalls that had created a 40 ha thermal plume in Biscayne Bay and was killing turtle grass (Langford, 1990). The bay is shallow with many areas under 3 m. Natural summer sea temperatures are 30-35°C to which a 10-15°C temperature increment (ΔT) was added. Four units (two oil/gas; two PWRs) are now cooled by a system of 32 warm canals, carrying water south, and eight return canals. There is no make-up apart from rain and ground water. The canals are about 60 m wide and between 0.3 and one m deep, separated by 27 m wide berms. The total length is about 270 km, giving an effective surface area of some 1560 ha (Figure 3.1). The design point for the canals was for a 110 m^3s^{-1} flow with a nominal 10°C ΔT. The water was described as "travelling on a two-day, 168-mile journey from canal to condenser". Table

3-2 shows the thermal loading of the Turkey Point canals to be at the higher end of the range for cooling lakes. However the relatively high and varying rate of flow (2-5 ms^{-1}) should give better air/water contact and evaporation than the often sluggish movement beyond the bunded areas of cooling lakes. Solar radiant heating, even in Florida, and heat loss to the ground are negligible compared to evaporative heat loss.

Table 3-2 Cooling capacity of Turkey Point (Fla) canal system

Installed capacity (Megawatts electrical) MWe	**Assumed efficiency %**	**Estimated heat rejection (Megawatts heat) MW**
coal/oil 2 units: 846 MWe	40	2,115
nuclear 2 units: 1,456 MWe	33	4,412
Total heat rejected		6,527 MWht

Surface area of canals	**Capacity ratio (MWe / unit area)**	**Cooling duty (MWht / unit area)**	**Rate of net heat loss**
1,560 ha	1.48 MWe ha^{-1}	4.18 MW ha^{-1}	418 $Wm^{-2}h^{-1}$

Figure 3-2 Aerial view of Turkey Point cooling canals

There is debate as to whether replacing 2,500 ha of mangrove swamp by canals was an acceptable substitute for the loss of turtle grass. On the other hand nearly one-quarter of the entire US crocodile population now lives and breeds in the “sanctuary” of the canals (Langford, 1990).

3.1.4 Spray ponds and assisted direct cooling

At this point we are reaching a cross-over. Having started with the sea, the subsequent cooling options have involved decreasing volumes of “available water” and have become increasingly “engineered”, culminating in cooling canals. There is no clear demarcation between a large cooling pond and a small cooling lake.

Cooling pond design principles

Kent (1938) estimated the relative ground areas required for dissipating a given heat load, as shown in Table 3.3.

Table 3-3 Relative areas required for cooling using different methods

Relative area	Wet cooling system
1,000	Cooling pond
50	Spray pond
15	Spray (unfilled) tower
4	Pack-filled natural draught tower
1.5	Mechanical draught counterflow tower
1 to 2	Mechanical draught crossflow tower

Cooling is primarily by evaporation and by convection from the pond surface; heat loss to the ground and solar heating are negligible compared to the evaporative heat loss. A relatively static pond can achieve heat rejection rates of around 23W $m^{-2}°C^{-1}$ in summer but, counter-intuitively, heat rejection actually can be lower in winter (10-12 W $m^{-2}°C^{-1}$). This is because, in low-humidity areas, 85-90 per cent of summer cooling is achieved by evaporative transfer, but in winter evaporation rates are far lower. The design point of a cooling pond is determined using maximum flow, maximum inlet temperature and maximum wet bulb temperature. Factors affecting cooling performance are depth, local topography, air temperature, relative humidity, windspeed and the amount of sunshine it receives. For a given heat load, higher pond operating temperatures will require a smaller area and incur less evaporative loss – a factor to be considered in arid areas. One factor that is sometimes overlooked is the retention time in the pond. The recool temperature of a pond with a short retention (under 24 hours) can fluctuate quite widely between day and night. Salinity, arising either from the use of seawater, saline aquifers or irrigation run-off, introduces other variables into the pond heat transfer equations: increased salinity reduces evaporation rate (for introduction see Calder and Neal, 1984; Thant Zin-win, 2004).

The primary cost item for a cooling pond is land and, unlike the cooling lakes described above, most have little or no aesthetic, recreational or environmental attraction, although some may be used as ornamental features – for example the (now demolished) Central Electricity Research Laboratories at Leatherhead used a sterile, blue-painted ornamental pond and fountain in front of the main entrance whilst Innogy (Swindon) uses landscaped vegetated ponds with fountains. Ponds are associated mainly with smaller-scale applications, such as cement works and steel and paper mills, where they may serve a number of requirements. They may be needed simply to reduce water temperature to the permitted level prior to discharge or for recycling for plant cooling, or for cooling high-temperature wastewater to under 35°C prior to biological treatment.

In some cases the pond itself has a purifying (water treatment) function, especially if aerated. As with lakes, pond inflow may be directed by bunds or partitions to maximise the cooling path.

The design point of a cooling pond is determined using maximum flow, maximum inlet temperature and maximum wet bulb temperature. A system that performs under these conditions needs no assistance but borderline cases can be improved by enhancing air/water contact by aerating (stirring) the surface or by using pumped sprays. If only a small, or occasional, cooling improvement is needed, a floating agitator/aerator could suffice. The capital cost of a spray cooler is about five times that of an aerator of the same horsepower but, in similar climatic conditions, has a heat dissipation rate some 10 times greater. This can cool to within 9-10°F (5.5°C) of wet bulb temperature. Sprays also enable the pond area to be reduced, since a pond equipped with spray coolers needs only five per cent of the area of a simple pond. Factors affecting spray cooling efficiency include surface area of the spray, relative velocities of air and water during contact, and time for which droplets are in the air (Titchenor, 1971). A spray cooling pond occupies 10-20 times more land than a wet cooling tower. However its capital costs are far lower, especially when uprating an existing static pond, which can be done using an off-the-shelf package of floating platform with pumps and multiple nozzle assembly. For large-scale applications, such as for the proposed 1970s Bradwell demonstration commercial high-temperature reactor, several hectares of seawater ponds with fixed spray heads were planned. The excavation spoil was to be used for site-raising.

Figure 3-3 Spray cooling (photo courtesy Siemens Power Generation)

3.1.5 Tower cooling - general issues

Tower cooling continues the progression towards more complex engineered structures and reduced water requirement. In wet towers, the air/water contact is enhanced by enclosing and directing the air flow and by maximising air/water contact by exposing the water as droplets (as in spray ponds) or as a thin trickling film. The pond beneath a wet tower serves to collect the falling water and its area is no more than the basal area of the tower. Cooling is by direct transfer of heat to air and by evaporation of some of the water. Evaporating one kg of water removes sufficient heat to cool 100 kg through 10°C; thus, as a minimum a wet tower station need abstract only one per cent of its circulating volume. In practice about three per cent is usually abstracted, of which one per cent is evaporative loss (consumption) and the other two per cent is discharged (purged) back to the waterway to control the build-up of dissolved and suspended solids. The trend of less water consumption continues via hybrid wet and dry systems, where the evaporative-cooling component is reduced, to dry towers where the condenser cooling water is in finned tubes and does not come into direct contact with air and thus neither benefits from, nor loses water to, evaporation. Ultimately, no cooling water is involved at all and the steam exiting the turbine is condensed in large air-cooled radiators.

For many years cooling towers were used only where insufficient water was available or readily accessible for direct cooling. In some cases this was seasonal and the towers were brought into use only in summer. However, tower cooling also permits more flexible siting, which can reduce fuel transport and/or transmission costs. Against this is the higher capital cost of towers, although these costs might be marginal when set against emplacing or tunnelling large diameter pipes for direct cooling. Towers, and particularly dry towers, have lower thermal efficiency because of the higher cooling water temperatures at the condenser inlet; this can reduce the station's efficiency by three to five per cent compared with direct cooling. The economics are not entirely negative, since with higher backpressure the dimensions of the turbine LP cylinders can be reduced, with fewer and smaller diameter discs and a corresponding reduction in the dimensions of the casing and exhaust steam passages. These economic arguments need to be weighed in each case. Furthermore, in the USA and in Europe, tower cooling is increasingly required for environmental reasons since less water is abstracted and less heat is discharged to the receiving waterway. It is for this reason that wet cooling towers, and even air-cooled towers and condensers, are being built in coastal locations. Many of the problems associated with cooling towers (chemical scaling, sliming and siltation) apply equally to fresh and salt water. Corrosion is more of a problem in saltwater towers and, whilst salt (NaCl) drift is unique to them, dissolved solids drift is an issue for any tower operating at high concentration factor (CF).

3.1.6 Tower components and terminology

Splash pack

Splash pack consists of tiers of horizontal, triangular-section, wooden or plastic slats wetted by droplets that form and reform as they shower down through them. Wooden laths, and other timber-supporting structures in the tower, have to be impregnated with chemicals (typically arsenic, chromium and copper mixtures) to prevent fungal rot and, in saline towers such as Connahs Quay A and Fleetwood, attack by burrowing gribble (the shrimp-like *Limnoria*). Chlorination and alkaline conditions can accelerate de-lignification, weakening the timber. Build-up of slimes, scale and silt on splash pack slats has been limited by their relatively small surface area and the incessant rain of water droplets. In some CEGB towers the pack extended below the "skirt" of the tower, where it was exposed to sunlight that encouraged a heavy growth of green algae (*Enteromorpha spp*.) that eventually broke the slats (Whitehouse and Coughlan, 1987).

Film-forming pack

Film pack has largely replaced splash pack. As the name suggests, water flows down its surfaces as a thin layer. The first film packs were individual sections of corrugated asbestos cement sheet but modern film packs consist of thin sheets of complexly corrugated, ribbed plastic (usually PVC) spot-welded to form lightweight blocks containing numerous vertical or zig-zag channels. The spacing of the component sheets (depth of corrugation) varies; there can be between 40 and 62 corrugated sheets per linear metre, providing a nominal surface area of up to 124 m^2 per cubic metre of pack. A combination of photogrammetry and topographical analysis showed that the patterns and ribbing increased this plane area by 30 per cent (Whitehouse and Coughlan, 1987). Towers designed for use with film packs have a water loading of around 2.1 kg m^{-2} s^{-1}, permitting a considerable reduction in size and cost. Recool temperatures of natural draught splash-pack towers retrofitted with film pack were between 2.5 and 4°C lower, even though the original water distributors and loading of about 1.25 kg m^{-2} s^{-1} were retained. These gains refer to clean, unfouled pack.

After installation, most film-pack manufacturers specify a 12-15 week "conditioning" period during which no biocides should be used. During this time the plastic loses its water-repellent shine, thereby improving its wettability and heat-transfer capabilities. These changes appear to be due to physical erosion, microbial action and chemical deposition. Thin slimes produced by the microbes initially improve recool but over time particles of silt and sand adhere to them and the whole becomes a complex matrix in which soft chemical scale and sediment serve as a reinforcing aggregate. A mature slime of this type is extremely resistant to biocide penetration and is almost impossible to remove, especially since chlorine is less efficacious in the alkaline conditions of most cooling circuits than in neutral or acidic water. In time these deposits will begin to slump and partially block the water passages. The combined weight of fouling and pent up water in the upper layers can deform or crush the channels at the base of the fill, resulting in yet further water retention and weight increase. In extreme cases the relatively flimsy structure supporting this "lightweight" pack collapses (Cottam power station, UK) but long before this point is reached the water distribution and air flow have been adversely affected - negating the efficiency advantages of, particularly, the more complex patterns of fill. In practice (a) the working life of the pack is often determined by biogrowth, rather than by degradation of the material and (b) the most thermally efficient pack - usually the most complex - is not necessarily the best option. In some situations a more open block or even a latticework "trickle fill" is indicated.

Purge or blowdown

Purge or blowdown is essential to prevent the accumulation of dissolved and, to a lesser extent, suspended solids in evaporative circuits. A high make-up and purge rate is not beneficial in high sediment waters and is uneconomic when water treatment chemicals such as acid and flocculants are being used. Suspended solids tend to settle out in tower ponds and at various other quiet points along the system and accumulate, rather than leaving in the purge. In time this can significantly reduce the volume of circulating water, which has knock-on effects since water chemistry is related to volume. It is prudent to factor in a value for potential loss of volume when estimating scaling potential. In extreme cases, particularly with intermittent make-up, loss of volume can lead to low-level trips of the CW pumps. In the commonly used one per cent evaporation/two per cent purge/three per cent make-up circuit, the concentration factor (CF) is 1.5 and a "particle" of water would, on average, go round the circuit 23 times before half of it had been either evaporated or purged (Table 3-4). This is sometimes referred to in terms of retention time; however, with continuous purge a fraction of the make-up will leave the system on its first circuit whilst some will still be in the system many months later. Make-up and purge both may be continuous or discontinuous. Discontinuous purge enables intermittently applied water treatment chemicals such as chlorine to be retained in circuit until they have decayed to a level suitable for discharge or until, for example, tidal conditions are suitable. Where temperature consents permit, it is thermally beneficial to purge water before it has been through a cooling tower.

Table 3-4 Relationship between concentration factor, half-life (retention) and make-up rate, assuming one per cent evaporative loss of the circulating volume

Make-up volume (%)	Purge volume (%)	Concentration factor	Half-life (cycles)
5	4	1.25	13.5
4	3	1.33	17
3	2	1.5	23
2	1	2	34
1.5	0.5	3	46
1	0.2	6	58

There are situations where even three per cent of the circulating volume exceeds the supply of water available. In such situations the CF must be increased; some stations in arid zones work on zero purge. High CFs and especially zero purge create difficulties with the chemical management of the circuit and continuous treatment is required to reduce the dissolved solids content to prevent heavy scaling in condensers and coolers. Cooling towers are very effective at scrubbing CO_2 from the circulating water, thereby raising the pH. This is usually redressed by adding acid[17] to keep the carbonate equilibrium on the (soluble) bicarbonate side, but can also be achieved by CO_2 injection, using scrubbed flue-gas. By coincidence, the use of treated sewage effluent as make-up reduces the need for acid since oxidation of ammonia yields nitric acid. An alternative approach to scale control is to soften all the incoming make-up water (as tried at Little Barford) or to continuously treat a fraction of the circulating water (side stream treatment). The sludge resulting from water treatment is almost impossible to dewater and typically is lagooned. A review of power requirements in the western United States concluded that most new build will be on inland sites. Here, they will be competing with agriculture and public users for water and will need to go down the high CF tower route. Environmental legislation will constrain them to dispose of the sludge in lined evaporation ponds (Jury *et al.* 1980).

Carryover or drift

Cooling water droplets of varying sizes are entrained in the rush of air emerging from the top of the pack but the majority of these should be removed by impingement against the "eliminators" where the air flow is forced to change direction rapidly (the same principle as in steam moisture removers). Not all droplets that have passed through the eliminators as carryover will leave the tower. Condensation can occur above the eliminators in tall natural draught towers and condensate and carryover

[17] UK stations tend to use sulphuric acid. This is more expensive than hydrochloric acid but is easier to handle since it does not fume in moist air.

droplets swirl around inside the tower ("uprain" and "downrain") growing in size through collisions and further condensation. Some droplets will be expelled from the top of the tower in the updraught but others become so large that they fall back onto the eliminators and drip back onto the pack. Much of the potential carryover is removed by this process. Mechanical draught towers are shorter and far less prone to internal condensation, and hence more likely to expel all carryover. Manufacturers usually specify a performance value for such non-evaporative losses but experience has shown that it is near-impossible to discriminate between carryover (drift), washout and leaks.

Washout or blowout

Water falling from the base of the pack into the pond may be blown out of the tower by crosswinds. Washout comprises droplets of "raw" cooling water and as such will carry dissolved and suspended solids and possibly pathogens. The concentration of suspended solids may be augmented by fouling washed out of the tower pack. Drift is mainly a problem with natural draught towers since these have a large unrestricted air gap below the shell through which falls a constant rain. On induced draught mechanical towers this gap is usually fitted with louvres to minimise through-wind, whilst on forced draught towers there is no gap since the fan is at the air entry. Typical drift values for natural draught towers are 0.3-1.0 per cent of the circulating rate and 0.1-0.3 per cent for mechanical draught towers although some manufacturers claim drift rates as low as 0.02 per cent and even 0.005 per cent if sufficient attention is paid to design and maintenance. With seawater towers, salt drift is of particular concern.

3.1.7 Wet towers

Configuration

There are three basic air/water combinations for both natural draught and mechanical wet towers. In co-current towers the air flow is downward, having been entrained by the descending water. In counterflow (countercurrent) towers the air moves upwards through the descending water. In crossflow towers the air moves laterally through the descending water. Co-current towers, also known as spray towers, are relatively short, have no moving parts and are the least efficient. Counterflow provides a more efficient heat transfer than crossflow, since the coolest water contacts the coolest air, but the towers are relatively taller with higher pumping costs. The water may fall freely as droplets (spray towers) but it is more usual to have some type of fill or pack to prolong the air/water contact time and to increase the area of the water surface, thereby improving the recool temperature. The warm water is distributed across the top of the pack by pans, channels or pipes fitted with nozzles or splash-type spray heads. For maximum efficiency there should be an even water loading and air flow across the entire pack. In circular-section natural draught towers, air flow tends to decrease at the centre but this can be countered by increasing the depth of pack around the edge.

Design principles

The key factors in tower design are the maximum water loading (inflow volume) and temperature, the air:water ratio and the wet bulb temperature. A close approach to wet bulb temperature (high temperature range) requires a large air:water ratio but, since airflow is the main determinant of tower size and cost, it cannot be increased economically beyond certain limits. The opposite strategy is to opt for a smaller, shorter tower with a small air:water ratio, a large approach and small range. Most design requirements fall somewhere between these extremes. Performance indicators and coefficients can indicate how close a particular tower comes to an ideal case but, in practice, such indicators are often meaningless since the most efficient tower is simply the one giving the best overall economy in its situation. To ensure consistent, adequate performance, a "worst case" wet bulb temperature (say one that would not be exceeded for more than one per cent of the time during the summer) is selected, based on historic data. Over the years, power plant cooling towers have been designed with approaches ranging from 5 to 12 °F (2.8°C to 6.7°C). Cooling to within 5°F of wet bulb temperature would give optimal performance but 8°F (4.5°C) is more typical.

Natural draught towers

In a natural draught tower, the tower shell (typically a hyperbolic profile concrete structure) extends high above the wet section and functions as a chimney, with the convective rise of warmed air drawing in cooler air at the tower base. Obviously this convective air flow will stop should the air temperature (dry bulb) exceed the temperature of the inlet water. However the tower does not cease to cool efficiently if the wet bulb temperature exceeds inlet water temperature. This is because the air is warmed as it enters the tower, thereby raising its wet bulb temperature. These towers do not require fans and have low operating costs but can incur maintenance costs. Moreover there is little if any scope for control and this often results in overcooling. The 1,000 MWe Ince B (UK) station was cooled by a large, fan-assisted natural draught tower. Previously a station of this size would have used four unassisted towers.

Mechanical draught towers

Mechanical draught towers, usually constructed of timber or plastic, are totally reliant on large axial flow fans to push or pull air through the pack, which gives a more consistent air flow than in natural draught towers. By using internal doors and variable speed or variable pitch fans, the performance of such towers can be matched to prevailing atmospheric conditions. In central and western USA, such towers are used exclusively since they can provide a more controlled performance in conditions ranging from freezing to hot and dry. In most situations, they provide lower recool temperatures than natural draught towers but their power consumption is typically about 0.5 per cent of the plant's output. Towers with the fan at the discharge (induced draught) have a low air entrance velocity but a high exit velocity that reduces the possibility of recirculating saturated discharged air back into the air intake. Forced draught towers, with the fan at the intake, have high entrance but low exit air velocities. The low exit velocity is much more prone to recirculation and icing. Teesside CCGT co-generation power station has a single hyperbolic tower with 22 fans around its base, giving an air flow of almost 10,500 kg s^{-1} and a water loading of 8,000 kg s^{-1}, to provide a cooling capacity of about 460 MWth. Depending on load and ambient conditions the inlet temperature averages 30°C with a 16.5°C recool. The station has two back-up forced draught hybrid towers to bring into use when the thermal export demand is reduced.

Hybrid tower systems

Many types of tower, or combinations of towers, can be described as "hybrid". Their common feature is that they all are used to cool water that has been used in a surface condenser.

(a) A natural draught tower augmented by fans at its base has a greater cooling capacity than a natural draught tower of similar size, and the fans can be used in response to load or season.

(b) A hybrid mechanical draught tower incorporates a "dry" (radiant) section above the wet pack. A proportion, or all, of the incoming water is routed through finned tubes in the dry section (sensible/dry bulb approach) before it is sprayed over the pack in the wet section (latent/wet bulb approach). In some cases this arrangement is adopted for "plume abatement"; one disadvantage of low profile mechanical draught towers is that any plume of vapour or drizzle is far closer to the ground than with a natural draught tower and this can cause ground fog and icing on nearby roads. The warm air coming through from the dry section mixes with the saturated air rising from the wet section, raising its dew point and giving added buoyancy to the plume. In some climates the tower can operate for much of the year using the dry section alone, greatly reducing annual water consumption.

(c) Hybrid pack is a type of plastic fill used in crossflow towers. Air flows through horizontal channels (pipes) in the matrix whilst water percolates over them, without direct contact. In an alternative design the wet and dry channels are inclined at 45° to the vertical. The aim is to create an extensive area of air-cooled surface over which the water flows in a thin film, with minimal evaporative loss. Both types of fill were trialled for Deeside CCGT (Coughlan, 1972) but were not installed owing to their high propensity for fouling. Salt and/or scale deposits can become a problem at the front face of some crossflow film packs. On the Deeside test-rig, seepage from the wet channels or recirculating spray intermittently wetted the front face. This rapidly dried, given the combination of a flow of dry air and warm pack, leaving encrusting deposits that accumulated and progressively restricted the air flow. The Rechem chemical plant (Fawley) had a freshwater tower fitted with this same pack and experienced a similar build-up of scale on the front face that was attributed wholly to seepage.

(d) Hybrid wet and dry cooling systems are an attempt to gain the advantages of both whilst offsetting the disadvantages of each – high water consumption and poor recool, respectively. They are essentially an extension of (b) above but use separate wet and dry towers (see Section 3.1.8). If designed for maximum water conservation, the design point is essentially an all-dry system with just enough wet capacity to prevent significant deterioration in recool during the hottest weather. This is sometimes referred to as a dry/wet peaking tower system. Alternatively the dry system could be used to improve the recool of a wet tower, either in parallel or series. In parallel cooling, some of the water from the inflow to the wet tower is diverted to the dry tower. This reduces the water loading on the wet tower, raises its air:water ratio and produces a lower recool temperature. The water leaving the dry unit is not as cold, but when mixed with the water in the tower pond, causes only a slight temperature increase. In series cooling, the water leaving the dry unit is fed back into the wet unit's inflow to reduce the tower inlet temperature, again with improved recool temperature. No working examples of these arrangements were identified. An Australian desk study (Williams, 2007) concluded that the additional power costs for pumps and fans on the dry unit would render retrofitting uneconomic.

(e) Helper or auxiliary towers are sometimes used to augment or completely replace the cooling function of a direct system during times of low river flow (Willington A) or when river temperature exceeds some preset value. Barking (lower Thames) is normally direct cooled but can switch to mechanical draught wet towers in summer; Didcot A (non-tidal Thames) has a stepped consent relating to river flow and temperature and, as a first step, uses a mechanical draught wet tower to cool the purge from its natural draught towers. The direct-cooled Brayton Point MA USA station, under pressure over fish protection and thermal discharge issues, plans to retrofit towers to cool 1,100 MWe of its plant, with the ability to use them as helper towers for the remaining 500 MWe.

3.1.8 Parallel Condensing System™ (PAC)

PAC is a patented GEA Group concept in which the exhaust steam from the turbine is split into two, variable, streams. One goes to a water-cooled surface condenser (with a wet-tower) and the other to an air-cooled condenser (See Section 3.1.10 below). Unlike the hybrid wet/dry towers described above, the dry unit of a PAC system is a condenser. Even on hot days the dry section can reject a substantial amount of heat, thereby reducing peak water usage. During the cooler months, if so designed, the heat rejected by the dry unit can be increased up to 100 per cent, with no evaporative losses. The plume can be reduced or eliminated entirely when danger of icing exists, simply by shutting down the wet section.

3.1.9 Dry towers (indirect dry cooling, fin-fan cooling)

Conventional surface condenser

The cooling circuit layout is the same as for a recirculating wet tower system except that when the water reaches the tower it does not come into contact with air but passes through finned tubes that are cooled by air flowing past them. There is no evaporative cooling, and hence no loss of cooling water, but the heat transfer to air via metal fins is much less efficient. The technology for small cooling applications is well established and usually comprises an elevated, horizontal cooling matrix with a horizontal fan beneath it. Duty can be increased by adding more modules. The largest installation is ESKOM's 4,500 MWe Kendal plant (South Africa). The 20 m^3s^{-1} CW flow from the condensers of each 686 MWe turbine passes through radially arranged, horizontal, heat exchangers (HEs) lying above the air inlet in one of six 165 m tall natural draught towers. The total length of finned, elliptical section, galvanised tube in each tower is 1,980 km, arranged in eleven isolatable sectors. The auxiliary power consumption/unit (additional pumps) is 3.4 MW (Du Preez, 2008). After 20 years operation the towers still match design performance and there has been no corrosion on the HEs.

Spray, jet or barometric condenser (or Heller system)

Spray condensers need no cooling water and no tubes. A fraction of the condensate is taken from the hotwell at the base of the condenser, cooled – usually with a natural draught dry cooling tower - and then sprayed back into the condenser to continue the condensation process. In 1961 a hyperbolic natural draught tower with vertical heat exchange matrices (radiators) around the air inlet was built at Rugeley A (UK) for a 192 MWe unit. The radiators were high purity aluminium to avoid internal corrosion and contamination of the circulating boiler feed water, (condensate) but the risk of external corrosion had been underestimated. Although the chloride content of the atmosphere was low, it became concentrated in crevices by evaporation and eventually caused corrosion and leaks. Several matrices were split by freezing. Since that time many systems have been built and operated successfully in Europe and worldwide.

3.1.10 Wet surface air coolers (closed-loop evaporative cooling, WSAC)

Dry coolers are totally reliant on sensible heat transfer (radiation) to air. Since air has a low heat capacity and the approach is restricted to dry bulb (air) temperature, the recool is always higher than for an equivalent wet cooling system.

Table 3-5 Economics of four methods of evaporative enhancement of dry coolers (adapted from Kutscher and Costenaro, 2002)

	Direct water contact	Up-front air cooling		
	Deluge over HE tubes (WSAC)	Coarse spray and wet eliminator pack	Fine spray misting system	Air drawn through wet pack
Total capital cost (ratio)	$37,139 (1.0)	$134,911 (3.6)	$155,977 (4.2)	$184,530 (5.0)
Additional cost /MW gain/year	$ 19	$ 34	$ 47	$ 60
Simple payback years	1	4	5	7
Internal rate of return %	165	32	23	16

Wet surface air coolers (WSAC) resemble conventional dry coolers in that the fluid to be cooled or condensed (see Section 3.1.11) is retained in a closed loop. However at the HE, water is cascaded over the cooler surface. Evaporation, aided by air draught, cools the HE tube surface and permits approach to within 5 to 10°F (2.8 to 5.5°C) of wet bulb temperature. Hauser (1982) estimated that wetting a “dry” HE increased its heat transfer three- to seven-fold. There are several interesting features in this approach (ref: Niagara Blower Company). WSAC has a far smaller footprint than an air-cooler, it consumes about 60 per cent less energy and delivers better recool. For example, on a 38°C dry bulb/24°C wet bulb day, the recool temperature of a dry cooler will be around 52°C when a WSAC could easily deliver 32°C. The water spray on the Niagara Blowers system is large volume/low pressure, typically delivering 4-7 $l\ s^{-1}\ m^{-2}$ on to the HE surface, in the same direction (co-current) as the fan-induced air flow. Consequently the sprays are accessible for inspection and maintenance. Excess water falls into the tower basin and air-borne droplets are separated and recovered as the air enters the fan plenum chamber. Make-up water can come from almost any source and can be used at high CFs since there is little potential for fouling as the HE tubes are not finned. Kutscher and Costenaro (2002) evaluated the benefits of four methods of evaporative enhancement of small dry cooling systems (Table 3-5). One was by WSAC deluge and the other three involved cooling the air upstream of the HE. The main advantage of the cooled-air system was that it avoided direct water contact with finned tubes; a problem that Niagara Blower avoids by using plain tube that, incidentally can be of almost any material and pressure rating. We could not find any example of WASC use on a large cooling circuit.

3.1.11 Air-cooled condensers (ACCs, direct dry cooling)

The indirect dry-cooled plant (Section 2.4.5) condensed steam in a standard surface (shell and tube) condenser and the cooling water was then circulated through a dry cooling tower. Direct dry cooling dispenses with the water loop and steam is exhausted directly to the cooler. However, the structures are necessarily large since the density of air is 830 times less than that of water and its thermal capacity is 75 per cent less. Thus Rye House (ref. Rye House), a UK CCGT station with an air-cooled condenser (250 MWe), has 600 A-frame radiator units mounted 24 m above the ground with 100 fans (6.1 m diameter, 68 rpm) below them. The radiators comprise 60,000 sections of oval finned tube, totalling more than 160 km in length.

No figures are available for Rye House but a 270 MWe steam turbine with a similarly sized ACC at Damhead Creek (Kent) has a design backpressure of 84 mbar that can rise to 150 mbar in adverse weather conditions, compared with a fairly constant 40 mbar for a typical water-cooled condenser (Turnpenny and Coughlan, 2003). Adverse conditions include high winds and gusts that can stall the fans, and little or no wind that can allow warm air to recycle. Overall, the efficiency of an ACC station is likely to be two per cent lower than a direct, water-cooled one. This equates to a loss of electrical output of at least 2.5 per cent, rising to four per cent (BREF/IPPC, 2000). On the other hand there is no evaporative consumption, abstraction or discharge of cooling water.

The largest ACCs are at ESKOM's 3,990 MWe Matimba plant (ref. GEA) with 48, 10 m diameter fans with a power consumption of 12 MW. Matimba has an average backpressure of 186 mbar with LP turbine protection at 650 mbar.

The external surfaces of ACC finned tubes are prone to fouling from pollen, dust, insects, leaves, plastic bags, bird and bat carcasses and so on, reducing airflow, heat transfer and performance and increasing operating costs. In severe cases, fouling can limit the power generation capacity (capability). Automatic washing/brushing equipment is often fitted (ref ConCo Services). Other problems include inward air leaks (the HEs are under vacuum) and freezing of the condensate. Aircraft engines, air-cooling systems and gas turbines all draw in large volumes of air and will catch insects and birds. The authors are not aware of any published literature quantifying this impact, but the parallels to aquatic impingement and entrainment are obvious (Veil *et al.* 2001).

Even in the UK, climatic conditions are such that several CCGTs with air-cooled auxiliary coolers have experienced operational problems during summer, despite their air-cooled condensers operating successfully. Rye House CCGT has an air-cooled condenser but uses wet towers to support critical functions such as cooling lubricating oil. Inadequate recool results in decreased viscosity and high turbo-alternator journal and thrust bearing temperatures and Paton *et al.* (2005) report that a few degrees change in oil temperature can alter the dynamic behaviour of the turbine shaft.

A recent study of the environmental impacts of wet and dry cooling (USEPA, 2000) concluded that energy consumption, per kg of condensate, was higher for dry cooling than for wet cooling and that the atmospheric emissions associated with that energy consumption also were higher. The energy penalty increases with the ambient air temperature. These disadvantages are offset by water demand being reduced by four to seven per cent compared with a recirculating system and 99 per cent over that required by a once-through system[18]. Dry cooling eliminates visual plumes, fog, mineral (salt) drift, cooling water treatment chemicals and several waste disposal issues. Nevertheless the study concludes that "*dry cooling does not represent the "best*

[18] Water demand here encompasses consumptive and non-consumptive uses and these may be derived from different sources. The one per cent remaining for the direct-cooled station would be for boiler feed and domestic use.

technology available (BTA)" for minimizing environmental impact" and EPA is concerned that "*the high costs and energy penalty of dry cooling systems may remove the incentive for replacing older coal-fired plants*" although this should not prevent ACCs being selected where there is no practical alternative. Sometimes wet towers, air cooling or ACCs may be built simply to forestall lengthy planning delays and environmental objections. One notable example is Mystic Harbour MA (EPRI, 2007), built on the edge of Boston Harbour. Air-cooled plants often use wet cooling for some auxiliary cooling purposes. Auxiliaries *are* air cooled at Crocket.

3.2 Issues on the use of seawater in cooling towers

3.2.1 Tower sizing

Water having more than 750 mg l^{-1} chloride (as NaCl) is considered saline, although full-strength seawater contains 26,000 mg l^{-1} NaCl and 30 to 34,000 mg l^{-1} total dissolved solids (3 to 3.4 per cent). With CF 1.3 to 1.5, the dissolved solids in the circulating water could reach five per cent or 50,000 mg l^{-1}. Seawater circuits seldom go above CF2, but some high CF arid-zone saline circuits will do so. At even 5,000 mg l^{-1} parameters such as specific gravity, boiling point/partial pressure and specific heat diverge significantly from freshwater values and must be allowed for in the design point calculations (see Maulbetsch & DiFilippo, 2008). Salinity increases specific gravity but, in thermal performance, the reduction in vapour pressure and specific heat (thermal capacity) outweighs this, so a saltwater tower must have a higher water flow or wider range to handle the same heat load. The thermal performance of a typical mechanical draught cooling tower could decrease by two to five per cent when operating at 50,000 mg l^{-1} dissolved solids. Operating at a lower CF would limit the build-up of dissolved solids, but at the risk of increasing the rate at which suspended solids are imported.

3.2.2 Scaling, corrosion and abrasion

Scaling and corrosion, often regarded as two sides of the same coin, must be considered at every stage of design for seawater circuits. The widely used Langelier Saturation Index is a useful indicator of carbonate scale formation but it is purely an equilibrium index and gives no indication of how much scale will form, or calcium carbonate will precipitate, in bringing the water to equilibrium. Scaling and corrosion are influenced by pH, hardness, alkalinity, total dissolved solids and temperature so in theory an equilibrium condition can be maintained so that neither occurs. However this tends to ignore the dynamics of the system. Seawater is naturally around pH 8 and is well buffered; this pH is outside the range 6-7 normally recommended for minimising scaling and corrosion. With low CF operation (relatively high purge flows) it is uneconomic and environmentally unacceptable to rely on dosing scale and corrosion inhibitors. Low volume circuits, where the aim is maintain high water velocities to keep silt and fine sand on the move, if not permanently in suspension, can abrade components and increase corrosion rates by scouring away potentially protective films.

3.2.3 Material selection

The principles for selecting and mixing component materials for use in seawater are fairly well understood. However, with tower cooling circuits and within the towers themselves, conditions are more extreme than those encountered routinely, since the temperature and salt concentration are elevated and the water can be supersaturated with air. It is a fact that corrosion proceeds most rapidly at an air/water interface (typically at the waterline) and in damp well-aerated conditions. In and around towers, alternate drying and wetting will first concentrate salts on a surface and then wash away potentially protective corrosion products (see Maulbetsch & DiFilippo, 2008).

For the tower ponds, and any internal structures to be fabricated in reinforced concrete, a minimal-water mix and the replacement of a proportion of the cement with 30 per cent pulverised fuel ash (PFA) or 60 per cent ground granulated blast furnace slag (GGBS) has been recommended. This reduces chloride diffusion in water temperatures up to 30° and 40° C respectively. The rebars themselves may be epoxy-coated but at minimum the bars should not be salt-contaminated from the atmosphere prior to placement. Steel should be avoided but pressure-treated timbers such as California redwood and Douglas fir are durable and there is no difference in their lifetime performance between seawater and freshwater towers. Controlled conditions and extreme care must be taken when applying coatings to steel or concrete on site. Coatings should be regarded as unreliable and best avoided if alternative corrosion-resistant materials are available. Medium-density polyethylene (MDPE) and glass-reinforced plastic (GRP) pipes and sections are preferable to coated steel. Even small items such as nuts, bolts and rivets may be subject to attack

The care taken in selecting materials for major items often does not extend to minor items. Thus at National Power's Killingholme CCGT (estuarine) the cooling towers' wooden internals had to be refixed within five years, replacing all the bronze fittings and hardware with a chromium/molybdenum steel (Xeron-100). Aluminium and silicon bronzes do not perform well in oxygenated saline waters. In particular silicon bronzes have poor resistance to erosion/corrosion, as caused by the impact of falling droplets, on the heads and nuts of through-bolts. They are also attacked by chlorine, particularly above 0.5 mgl^{-1}. With this experience National Power insisted on Xeron-100 (cheaper Xeron-25 was not suitable) for the towers at Deeside CCGT (estuarine, commissioned around1994). These towers had an upper dry section for plume abatement that used 90:10 Cu:Ni tube, a relatively cheap material. However, these tubes suffered rapid and severe internal erosion/corrosion and were soon replaced by titanium, which would have seemed the logical initial choice since titanium tube had been specified for the condensers. PowerGen's Connahs Quay plant, across the river from Deeside, uses the same Cu:Ni tubes in its dry sections but has experienced no problems. This suggests that factors other than water chemistry were involved and it is possible that sand scour was responsible for the severity of the damage at Deeside.

3.2.4 Purge (blowdown)

Purge or blowdown from a saline tower necessarily has a higher concentration of solids than the source water. On a seawater system, where there should be no shortage of make-up water, purge is unlikely to exceed double strength and, given the relatively small volume, it should not pose any regulatory problem other than the usual concerns over biocides.

3.2.5 Salt carryover

The plume emerging from the top of a tower should be almost pure condensate and as such carry virtually no salt if the eliminators are of good design, are all in place and the air and water loadings are correct. This situation can change rapidly should the water distribution alter as a result of blocked distributors or there is distorted or displaced pack. If a pack section is missing, the increased updraught can dislodge the eliminators enabling significant carryover of droplets. However, the main causes of damage to eliminators were attributed to careless handling and/or refitting following water-distributor maintenance. It is difficult to measure how much water is being lost as carryover and so it is generally included with drift loss, that is, all cooling water loss other than by evaporation. Since the total loss includes leakage and the evaporative loss itself is obtained by calculation from the cooling duty, any estimate of carryover is prone to considerable error. Carryover, especially from natural draught towers, is ejected at high level and as the droplets are carried away from the tower they diminish in size by evaporation and become less liable to reach the ground before they have become widely dispersed.

3.2.6 Salt water washout

Drift is water that is blown sideways from the base of cooling towers. This is a particular feature of natural draught towers that have a large gap (air entry), but can also affect mechanical draught towers in strong crosswinds or if the louvres are inadequate. Drift is mainly relatively large droplets and, starting near ground level, seldom spreads far off site. Effects on the cladding and structure of buildings, transformers and switchgear, gas turbine air intakes and even landscape planting should be considered at the design stage. Marley (1986) recommends that even though low levels of drift are attainable – under 0.005 per cent is cited - a saltwater cooling tower should never be sited near sensitive equipment. Apart from dissolved salts, these drops will also contain silt particles carried by the circulating water, possibly augmented by solids washed off the tower pack. The environmental aspects of salt drift and carryover are considered in Section 6.2.1.

3.2.7 Sedimentation

This problem is generally no greater at coastal sites than at many lowland river sites; in fact, marine sediments are often coarser than river sediments and are more easily removed from the incoming water. If left, coarse and fine sand will almost certainly drop out in the circuit. At Deeside CCGT, the turnkey contractor was unwilling to modify the design despite evidence of high concentrations of suspended sand. The contractor insisted that sedimentation would be controlled by maintaining high water velocities through the plant and having no pond (just a sloping tray) under the towers. Eventually the intake design was modified to provide a small, largely ineffectual, sand trap and, after numerous problems, hydrocyclones had to be installed at the intake. At Connah's Quay, across the river from Deeside, the incoming water is pumped into a large settling basin with an automated sand-removal system. This problem would clearly be greater on macrotidal water such as the Severn Estuary/Bristol Channel, where suspended sediment concentrations may reach 5 g l^{-1}.

3.2.8 Biofouling

There are two broad categories of biofouling - microbiofouling and macrobiofouling. The former comprise bacterial and fungal slimes that begin to develop on virtually any wetted surface - fresh or saltwater - within minutes of immersion. The most immediate effect of slimes is the reduction of heat transfer in condensers but they can also accelerate corrosion (microbiologically induced corrosion MIC) of metals and concrete. Condenser slimes never used to be a problem at seawater-cooled UK power stations and this was thought to be due to the surface toxicity of copper in the tube alloys, possibly aided by sand abrasion. The introduction of titanium tubes that lack inherent toxicity allowed slimes to develop, so most new stations specify a mechanical tube-cleaning system. The ability of slimes to incorporate sediments that greatly increase slime thickness, strength and weight is referred to in Section 3.1.6. Investigations into the feasibility of incorporating copper or some other slow-release biocide into plastic pack concluded that this was practically, economically and environmentally unsound. The relatively uncontrolled leach-rate meant that the biocidal effect would be exhausted within two to three years, far short of the intended pack life. Fungi also are implicated in the biodegradation (rotting) of timber; some timbers are naturally resistant to rot, at least in the short term. Others lack this inherent chemical protection and should be pressure treated with a proprietary compound.

3.3 Reactor cooling and ultimate heat sinks

For nuclear stations, routine cooling is required for reactor-associated equipment. In the UK this is referred to as reactor cooling water RCW or, more accurately, essential services water ESW. Apart from some early Magnox stations, this is provided by small, dedicated CW pumps and pipework. Over the years there has been increased physical separation of the main cooling water system MCW and “safety critical” ESW circuits. When a reactor is shut down these ESW circuits are still required and must continue to operate, even if there is no external electricity supply. Design must also take into account the worse case accident and potential loss of the usual water source for ESW.

Gas/graphite Magnox and AGR reactors have a large core and a low power density. Heat is transferred to the “boilers” by carbon dioxide gas at 20 to 40 bar pressure. The reactor core and boilers are contained within the reinforced concrete pressure vessel. Short of pressure vessel failure, any leakage of carbon dioxide coolant would be via one of the relatively small penetrations. Decompression would be relatively slow with no disruption of the graphite moderator and fuel geometry. Heat removal would be no more onerous than during a normal shutdown. By contrast in a PWR and BWR the water surrounding the fuel elements serves as both coolant and moderator. It is liquid only because it is under pressure (155 and 75 bar respectively). A major break in the PWR primary circuit or BWR steam circuit would see most of the water in the reactor flash explosively to steam (LOCA – loss of coolant accident) with potential overheating and distortion of the fuel assemblies. LOCA brings into play a number of flooding and re-circulatory cooling functions that are not part of a routine reactor shutdown.

The Ultimate Heat Sink UHS is required to remove the heat still being generated by the reactor core and in some scenarios the containment building, and various other pumps, circulators and diesel generators. At Sizewell PWR the UHS is a large air-cooled radiator servicing a water/water heat exchanger in the reactor area. Lochbaum (2007) shows UHSs at Limerick (Pennsylvania) and North Anna (Virginia) that use a pond with water sprays; Grand Gulf (Mississippi) that uses mechanical draught towers; Vermont Yankee that uses two dedicated cells in one of its two MCW mechanical draught towers and the South Texas Project that uses a small U-shaped pond. The South Texas Project pond design is detailed by Struble (2009). However configured, the UHS is supposed to provide all of the nuclear power reactor’s cooling water and make-up water needs for the first 30 days of an accident.

4 CW system design

4.1 Direct CW systems

4.1.1 System description

Direct CW systems usually require a coastal location. Figure 4-1 shows a schematic arrangement of a direct CW system with an inshore intake and an inshore outfall.

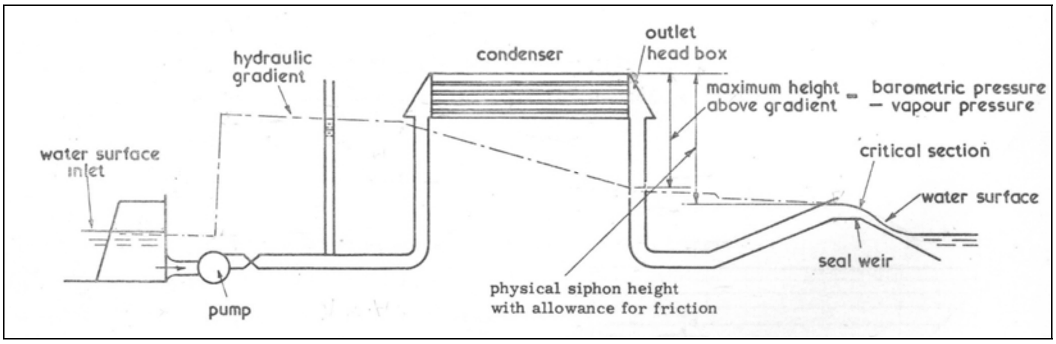


Figure 4-1 Diagram of typical direct CW system

The main components of the system are:

- intake structure;
- pumphouse;
- onshore inlet pressure conduit;
- condenser;
- onshore outlet pressure conduit;
- seal weir;
- outfall structure.

Where a CW system has an offshore intake, there is an offshore intake conduit between the intake structure and the onshore pumphouse. Similarly, where a CW system has an offshore outfall, there is an offshore outfall conduit between the seal weir structure and the offshore outfall structure.

Figure 4-1 also shows the hydraulic gradient of the CW system, defined as the level to which water rises in a tube connected to the system wall. Figure 4-1 illustrates two important features of a typical direct CW system. Firstly, to minimise pumping costs, parts of the system operate with a partial vacuum because they are above the hydraulic gradient. Hence a direct CW system is often referred to as a syphonic CW system. The system therefore has to be primed as part of the start-up procedure. Secondly, the system usually incorporates a seal weir which is open to atmospheric pressure. The purpose of the weir is to ensure that the hydraulic gradient cannot fall below the system by more than the maximum allowable siphon leg, otherwise the siphon would be broken. Miller (1971) discusses the hydraulic design of syphonic CW systems.

The height from intake water level to seal weir level is referred to as the ‘static lift’ of the CW system. If the intake is tidal, the static lift and hence the CW flow vary with tide level. Often the seal weir only comes into effect at low tide. At high tide the seal weir can be drowned, in which case the CW flow remains constant because the static lift has been eliminated.

4.1.2 Design philosophy

The fundamental function of a direct CW system is to provide a reliable supply of relatively cold water to the condenser. As the intake and the outfall are normally located in the same body of water they need to be separated so that the rise in ambient temperature at the intake, or ‘recirculation’, is minimised to an acceptable level.

Separation can be achieved vertically, horizontally or by a combination of both. Vertical separation can be used if there is natural stratification within the source of cooling water or when the intake is sufficiently deep to avoid ‘draw-down’ of the surface warm water plume from the outfall.

The design of horizontal separation parallel and perpendicular to the shoreline depends on site-specific factors such as the proximity to deep water, the strength and direction of the current as modified by the tidal stream, and the availability of any natural coastal feature that can be used as a barrier. For example, the intake and outfall for Heysham A are only 215 m apart as the crow flies but they are separated by the sea wall of Heysham Harbour which is 490 m long.

4.1.3 Design process

The design of a typical direct CW system for a given location may be subdivided into the following stages:

- determination of the CW flow and temperature rise from the condenser design;
- preliminary selection of the intake and outfall positions based on available published maps, charts and data on tide levels and currents;
- preliminary assessment of the likely ground conditions;
- preliminary design of the general layout of the CW system including intake and outfall structures, number and size of offshore and onshore conduits, and pumphouse plant arrangement;
- preliminary assessment of the system hydraulic gradient;
- preliminary assessment of heat dispersion at the outfall and the environmental impact of the system;
- surveys to obtain site-specific data on ground conditions, bathymetry, tide levels, currents, silt content, waves and water chemistry;
- detailed assessment of the heat dispersion at the outfall and the environmental impact of the system;

- detailed structural design of the CW system components culminating in the set of construction drawings.

The design is an iterative process. The number of iterations depends on the data available and acceptability of the proposed design in terms of heat dispersion and environmental impact. For example, the general arrangement of the outfall may need to be changed to achieve the desired heat dispersion.

The timing of the various design process stages is linked to the phases of the development of the power station. For example, the design concept and environmental impact need to be established for inclusion in the Environmental Statement. On the other hand, the structural design of individual system components may not be finalised until the construction phase. The surveys are often split. A hydrographic survey is usually carried out as part of the pre-application studies. The detailed ground investigation is often carried out at the start of the construction phase, the choice of conduit type being left to the construction contractor.

4.1.4 Optimisation

Ideally the construction cost and operating cost of a direct CW system should be optimised.

The optimum size of the onshore and offshore conduits should be determined by estimating the construction cost and the whole lifecycle CW pumping cost for a range of possible conduit sizes. Usually the optimum velocity lies in the range 3.0 to 3.5 ms^{-1}.

Subject to any overriding environmental considerations, the relative positions of the intake and outfall should be optimised to minimise the combined total of the construction cost, whole lifecycle CW pumping cost and cost of recirculation.

Assuming the system design requires a seal weir, the level of the site should be optimised to minimise the combined total of the cost of earthworks to lower the site and corresponding whole lifecycle CW pumping cost.

However, optimisation is not always possible because of site constraints. For example, the site level may be governed by flood defence or vehicle access requirements.

4.1.5 Design considerations

Inshore intake structure

An inshore intake requires a relatively sheltered location. This can be provided by an existing coastal feature such as a harbour or a loch (Figure 4-2) or a haven built as part of the power station project (Figure 4-3). In addition to environmental impact, the design of an inshore intake needs to take into account the requirements of local shipping, maintenance dredging and the impact on coastal processes such as the longshore transport of littoral drift.

Figure 4-2 Inshore Intake at Ballylumford B, Larne

Figure 4-3 Inshore Intake at Kilroot, Carrickfergus

Offshore intake structure

Prior to 1970 the majority of offshore conduits were low level tunnels with vertical shafts at either end. The offshore intakes were generally close to the sea bed. A permanent access facility was provided for the isolation, inspection and maintenance of the intake tunnels as well as maintenance of the intake coarse screens (Figure 4-4). Since 1970 offshore intake structures have tended to be capped, radial flow structures that project above the sea bed without permanent access for maintenance (Figure 4-5).

Intake tunnels have been found to be extremely reliable and the need to isolate them has been minimal. Currently the use of modern floating equipment is preferred for intake maintenance over the cost of providing a permanent access facility which itself is costly to maintain.

Figure 4-4 Offshore intake and inshore outfall at Wylfa, Anglesey

Figure 4-5 Capped radial flow intake structures for Vasilikos, Cyprus

The design of an offshore intake structure with respect to its position needs to take account of:

- the depth of water required to avoid the formation of surface-piercing vortices at low water;
- the depth of water required to avoid recirculation resulting from drawdown of the outfall plume (Miller and Brighouse, 1984);
- interaction with other CW systems, future as well as present;
- interaction with other industrial discharge outfalls;
- local shipping and fishery requirements;
- wave, tide and current conditions (for example, currents are weaker in bays than off headlands);
- suitability of the ground conditions for construction of the works;
- the presence of depressions in the sea bed that could collect debris such as dead seaweed;
- the presence of steep slopes in the sea bed where instability could result in slides.

The design of an offshore intake structure with respect to its structural form needs to take account of:

- the need to provide a coarse outer bar screen;
- the need to limit the intake velocity through the bar screen to mitigate the number of fish drawn into the intake;
- the need to provide an intake sill which is at least 1.5 m above sea bed level to minimise the amount of sediment and seaweed drawn into the intake, as well as epibenthic fish;
- the impact of debris such as fishing nets, ropes and plastic items;
- possible collision with a ship or a ship’s anchor;
- possible wave slam forces if the intake cap is exposed in the trough of a very large wave;
- the layout of the diffuser for the biocide-dosing system;
- the proposed method of access for periodic maintenance.

Offshore intake and outfall conduits

Up to 1970 the majority of offshore conduits were deep tunnels with *in situ* or bolted segment linings (Figure 4-6). In 1978 the first offshore conduits in the UK were constructed at Kilroot using the immersed tube technique as an alternative to tunnelling (Figure 4-7) (Carvell and Roberts, 1982). The method consists of placing precast concrete units in a shallow trench dredged in the sea bed. Subsequently, the method was used at Sizewell B and South Humber Bank (Barratt and Hamlin, 1998).

In the UK the tendency has been to use precast concrete units up to 94 m in length for economies of scale and to minimise the number of joints that have to be made underwater. Elsewhere immersed tubes have been built using 10 m long interlocking precast concrete units, such as Black Point in Hong Kong and Manjung in Malaysia. GRP pipes can also be used where conditions are good for diver work, such as Vasilikos in Cyprus.

The choice between tunnelling and immersed tube depends primarily on the ground conditions, environmental impact of construction, feasibility of floating precast units to the site, cost and contractor's preferred method.

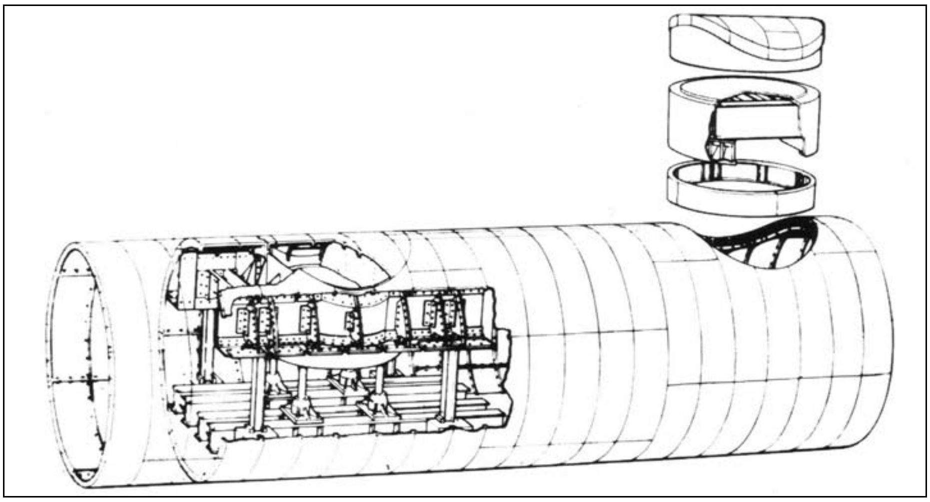


Figure 4-6 Bolted tunnel and shaft lining for Sizewell A

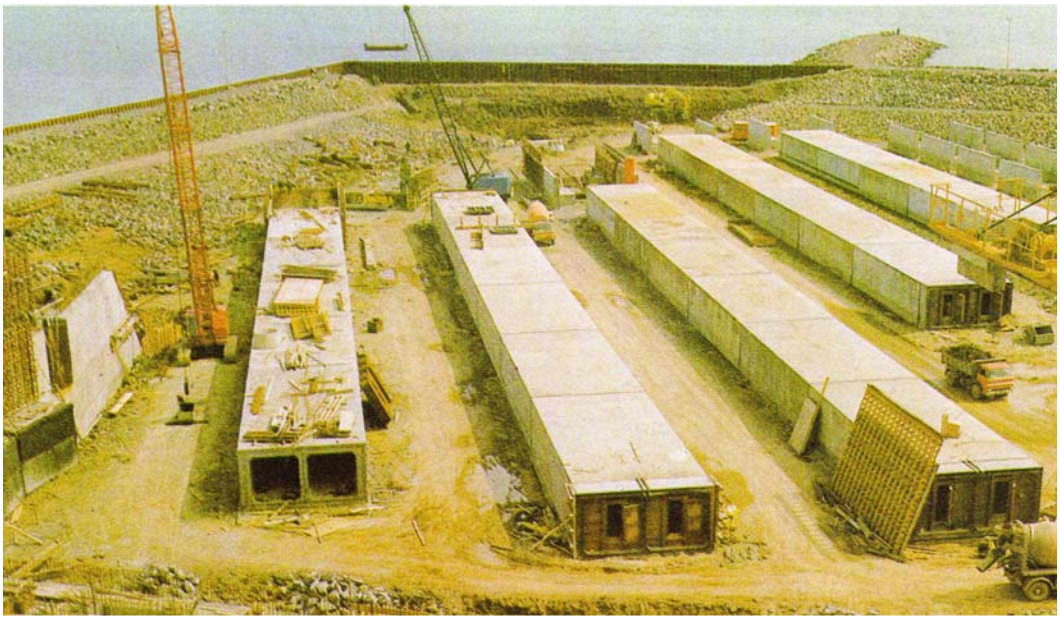


Figure 4-7 Precast offshore outfall conduits for Kilroot

Pumphouse

The CW pumphouse for a large power station is a major structure requiring substantial temporary works (Figure 4-8).

The plant accommodated at the pumphouse includes two rows of gates for double isolation of the plant, bar screens, fine screens, trash handling system, fish bypass, main CW pumps, auxiliary CW pumps, fire fighting pumps, CW pipes and valves, and overhead cranes for screen and pump maintenance. The bar screens can be mechanically cleaned. The fine screens can be band or drum screens (see Section 5.5)

The layout and number of fine screens and main CW pumps per generating set is determined by the plant designer. The screens are usually open air and the main CW pumps are usually covered by a superstructure.

In order to minimise hydraulic losses through the pumphouse, the forebay must be deep enough for the inlet system to flow fully submerged at extreme low water and the fine screens must be designed to pass the design flow at extreme low water. The main CW pump suction chamber must also be sufficiently deep for the pump submergence to be great enough to avoid vortex formation at extreme low water.

Figure 4-8 CW pumphouse substructure under construction at Vasilikos, Cyprus

Onshore inlet and outlet pressure conduits

The number of onshore conduits depends on the number of condensers, whether or not the main CW pumps and the condensers are 'unitised', and the outcome of the optimisation exercise referred to earlier. Typical materials for shallow conduits are reinforced concrete, pre-stressed concrete, steel and GRP. The Bonna pipe[19] is a composite pipe comprising a steel lining supported by an external reinforced concrete pipe and protected by an internal mortar lining. If the ground conditions are suitable for tunnelling, the options for deep conduits are either circular or horse-shoe section tunnels.

[19] http://www.stanton-bonna.co.uk (viewed 02/03/09)

Condenser

The detailed design of the condenser is determined by the manufacturer. However, the design and operation of the CW system is closely linked with the condenser. In particular, air inlet valves generally need to be incorporated in the CW system immediately downstream of the condenser. The purpose of these valves is to admit air into the system to prevent the development of high transient pressures in the event that the main CW pumps trip. The required capacity of the valves is designed by carrying out a dynamic analysis of the entire CW system.

Seal weir structure

The layout of the seal weir structure must ensure that the CW flow over the weir is uniform so that the hydraulic gradient can be predicted reliably.

There is a risk that air can be entrained in the CW flow in the outfall system if there is a large difference between the seal weir crest level and low water level. Entrained air does not generally affect the performance of inshore surface outfalls because it is safely released to the atmosphere. However, entrained air can cause problems in offshore outfall conduits depending on the layout of the system.

At Hunterston B, extreme low tide is 6.5 m below the crest of the seal weir. The offshore outfall conduit is a deep tunnel with a vertical shaft to connect it to the seal weir structure. The entrained air triggered an unstable surface oscillation in the shaft. Air collected in the low level tunnel and released itself up the shaft, resulting in shock pressures on the shaft wall. These problems were overcome by retrofitting a system that collected and released the air in a controlled manner (Miller, 1973).

Where shallow offshore outfall conduits slope down towards the outfall structure, pockets of air can migrate against the direction of flow and be released in the seal weir structure. Usually, this periodic release of air does not cause a problem.

Inshore outfall structure

The function of an inshore outfall structure is to convey the CW flow across the foreshore and to discharge it as a surface plume (Figure 4-4 and Figure 4-9). In addition to environmental impact, the design of an inshore outfall needs to take into account the requirements of local shipping and the impact on coastal processes such as the longshore transport of littoral drift.

Figure 4-9 Inshore outfall structures for Heysham A and Heysham B

Offshore outfall structure

An offshore outfall structure must satisfy two functions. Firstly, the outfall must discharge the CW flow in such a way that it satisfies the water quality standards set in the discharge consent. Secondly, the outfall must discharge the CW flow in such a way that the resultant surface plume remains as far away as possible from the intake at all states of the tide.

If excess head is available at the seal weir, this can be used to form a horizontal jet directed away from the intake (Figure 4-10). A more typical layout is to have a diffuser arrangement to maximise the initial dilution of the CW discharge.

The design of an offshore outfall structure with respect to its position needs to take account of:

- the need to avoid recirculation;
- interaction with other CW systems, future as well as present;
- interaction with other industrial discharge outfalls;
- local shipping and fishery requirements;
- wave, tide and current conditions (for example, currents are weaker in bays than off headlands);
- suitability of the ground conditions for construction of the works.

The design of an offshore outfall structure with respect to its structural form needs to take account of:

- the need to achieve the initial dilution set in the discharge consent;
- the need to provide a cap to prevent debris falling into the outfall;
- the need to provide a coarse outer bar screen for safety reasons;
- possible collision with a ship or a ship's anchor;

- possible wave slam forces if the outfall cap is exposed in the trough of a very large wave;
- the proposed method of access for periodic maintenance.

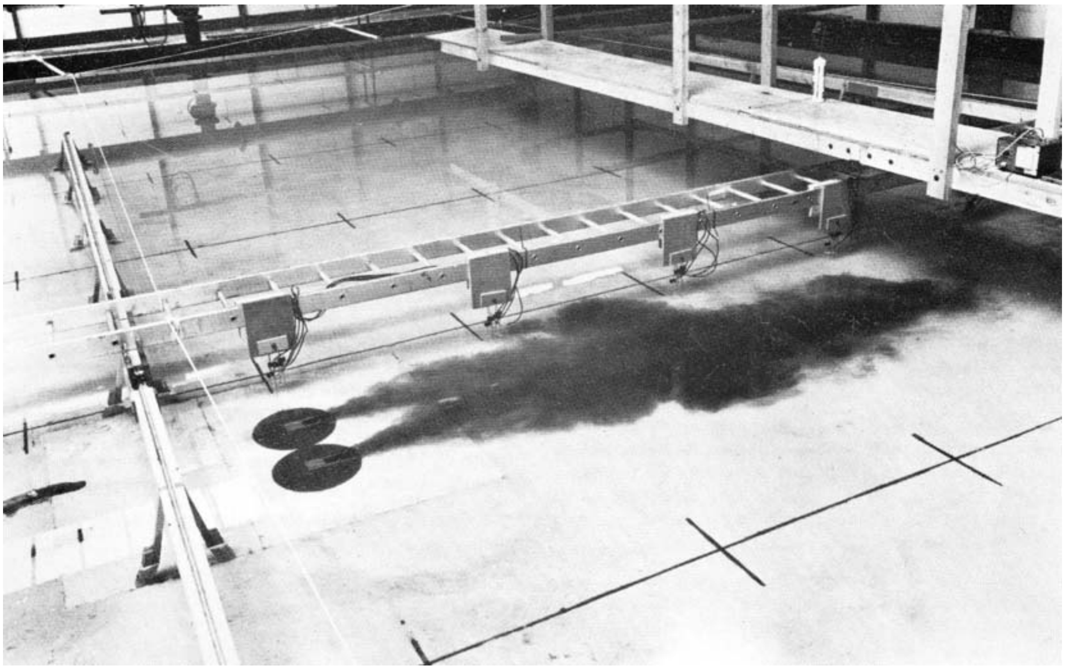


Figure 4-10 Model of submerged outfall for Kilroot

4.2 Indirect CW Systems

4.2.1 System description

Figure 4-11 shows a schematic arrangement of a typical indirect CW system. The cooling water is pumped from the main CW pumphouse through the condenser to a heat exchanger where it is cooled before it flows under gravity back to the main CW pumphouse. Normally the heat exchanger is a cooling tower which works on the principle of evaporation (see Chapter 2). About one per cent of the main CW flow is transferred to the atmosphere in the cooling tower. A further two per cent of the circulating CW flow is drawn off the system downstream of the condenser in order to prevent the build-up of contaminants in the CW system. The total system losses of about three per cent of the main CW flow are replenished by a make-up water system which pumps water from the primary CW source to the pond under the cooling tower. The purge water is returned to the primary CW source at a point which is remote from the make-up water intake.

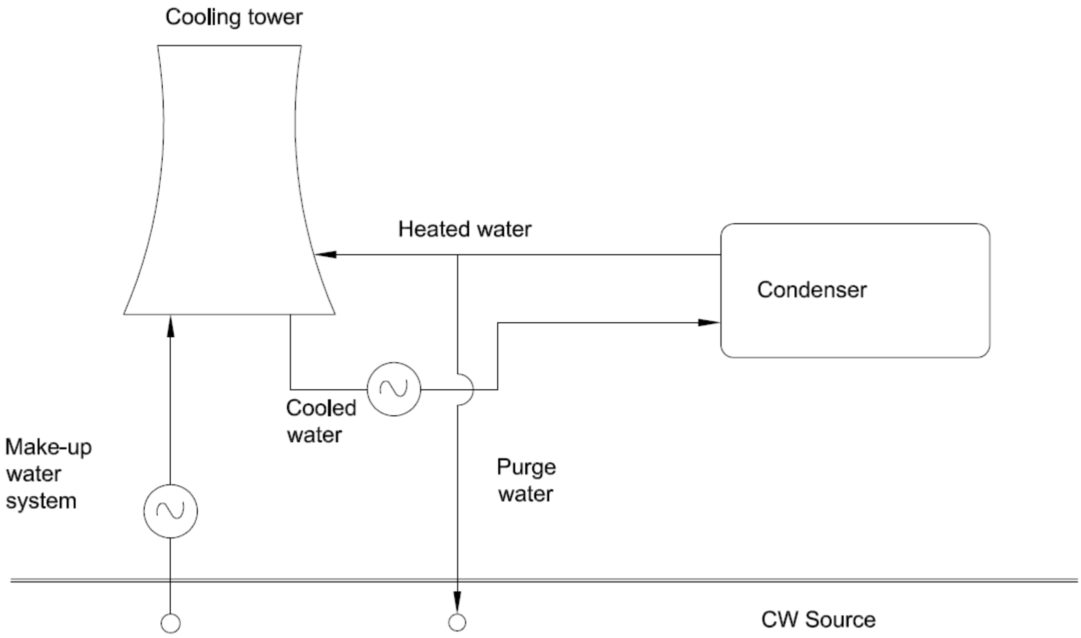


Figure 4-11 Diagram of typical indirect CW system

4.2.2 Types of cooling tower

Large evaporative cooling towers for power stations may be categorised by type (natural draught, mechanical draught, and hybrid) and characteristics (flow configuration, shape and construction).

There are two classes of flow configuration; counterflow and crossflow. Counterflow is where the flow of circulating water in the tower is in the opposite direction to the flow of air. Crossflow is where the flow of circulating water in the tower is perpendicular to the flow of air.

Other specialised tower designs have been developed to meet particular conditions.

Natural draught towers

A typical natural draught cooling tower comprises a hyperbolic shell structure mounted on leg supports over a circular pond. The circulating water enters the tower and cascades over a slatted structure, referred to as the 'packing', mounted above the pond. The packing is the primary heat transfer surface in the tower. Figure 4-12 illustrates the counterflow and crossflow configurations. The draught is driven by the difference in density of the warm moist air inside the tower compared with the cooler air outside the shell. The towers are extremely dependable because the draught is generated by a natural phenomenon.

The hyperbolic shell structure is larger and more expensive than mechanical draft towers. However, a natural draught tower is economic over the life of a station owing to the absence of mechanical equipment.

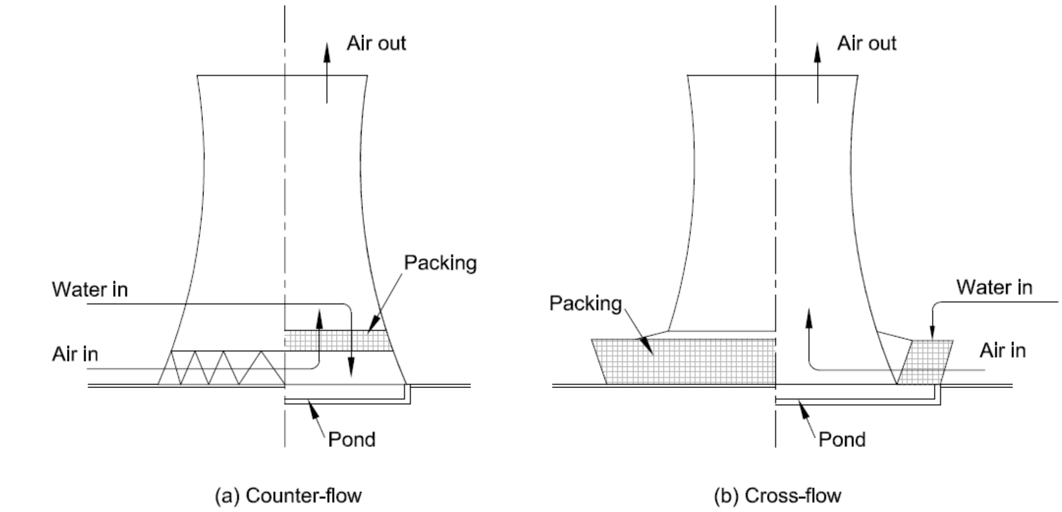


Figure 4-12 Configurations of natural draught cooling tower

Mechanical draught towers

Mechanical draught cooling towers are much smaller than natural draught towers. The mechanical draught required for evaporative cooling can be provided in two ways; a forced draught or an induced draught. The forced draught is generated by fans situated at the air inlet to the tower. The induced draught is generated by fans situated at the air outlet of the tower. Forced draught towers are prone to recirculation because they have high air entrance velocities and low exit velocities. Also forced draught fans can become subject to icing when moving air laden with moisture. Induced draught towers are therefore preferred for power stations. Figure 4-13 illustrates the counterflow and crossflow configurations for an induced draught tower.

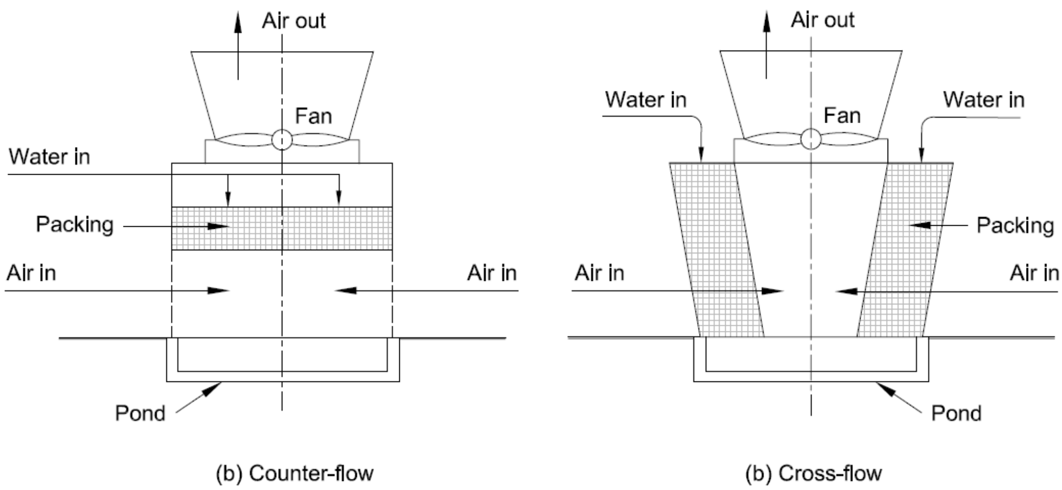


Figure 4-13 Configurations of induced draught cooling tower

The air entrance velocity for an induced draught tower is typically 2 $ms^{-1}$and the exit velocity is typically 6 to 8 ms^{-1}. The risks of recirculation and ice formation are substantially less than for a forced draught tower.

Mechanical draught towers can be arranged so the footprint is either circular or rectangular to suit the available site. Many of the components can be fabricated in the factory and the amount of on-site construction work can therefore be minimised.

Hybrid towers

A hybrid tower is a short natural draught tower to which induced draught fans have been added to augment the air flow, Figure 4-14. These towers are also referred to as fan-assisted natural draught towers.

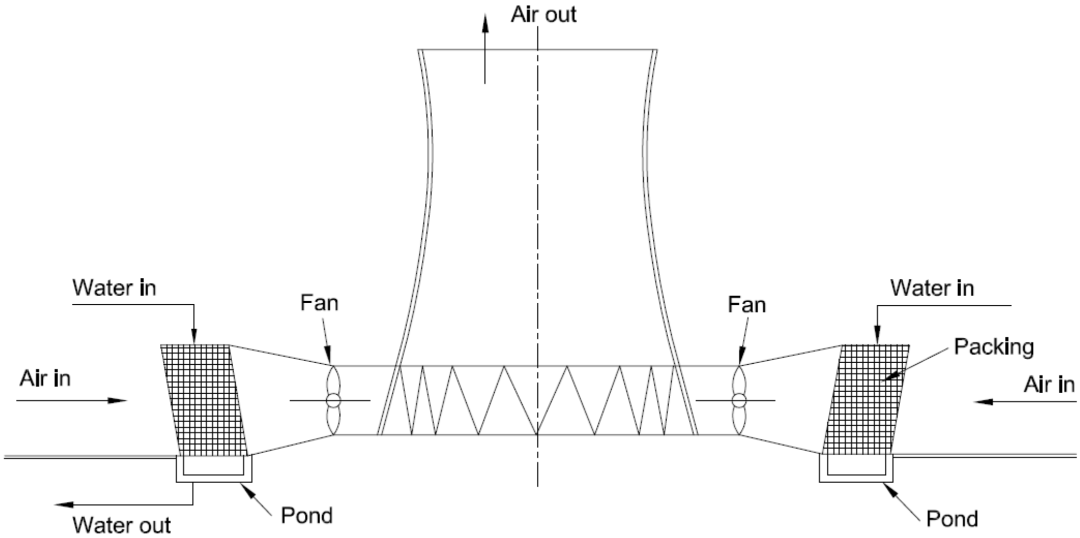


Figure 4-14 Typical hybrid tower arrangement

The purpose of the design is to optimise the cost of running the fans and the cost of constructing the hyperbolic shell of the tower. The hybrid tower can be advantageous when the relatively low level of the discharge plume from a conventional mechanical draught tower is not acceptable. Sometimes the fans only need to be operated during periods of peak load.

Other specialised towers

Dry induced draught towers can be used where make-up water is in critically short supply. The circulating water is passed through finned tube heat exchanger sections for the transfer of heat to the atmosphere. This process depends solely on the dry bulb temperature of the entering air. Consequently the cooled water temperature achieved is typically 11 to 17°C higher than that for a normal evaporative cooling tower.

Water conservation towers can be used where make-up water is available but scarce. This design comprises a group of dry towers linked to an evaporative induced draught tower. The circulating water is passed through the dry towers first. If additional cooling is required, the flow is directed to the evaporative induced draught tower. Conversely, if the temperature of the air-cooled water is adequate the water bypasses the evaporative section of the tower.

Conventional cooling towers produce a highly visible plume because the air leaving the tower is supersaturated. The plume abatement tower is designed to reduce the density and persistence of the plume significantly. The design of the tower consists essentially of using a combination of dry and evaporative cooling and mixing the exit air from both processes. The resultant plume is less visible because only part of the plume is supersaturated. The degree of plume abatement depends on the ambient air characteristics and the ratio of dry to wet cooling.

In certain circumstances, plumes from conventional induced draught towers can return to ground level and cause fog-like conditions downwind of the tower. This problem can be avoided by increasing the height of the tower by extending the 'funnels' above the fans or by increasing the height of the tower structure itself.

Spray-filled towers are towers in which the packing has been omitted. They are used where contaminants or solids in the circulating water would jeopardise a normal heat transfer surface. Their use is normally limited to those situations where higher circulating water temperatures are permissible.

4.2.3 Sources of make-up water

The source of make-up water for a large power station can be virtually any body of water, provided it can sustain an abstraction rate of about 2 m^3s^{-1}.

Other less likely sources are groundwater and the effluent from water treatment works.

4.2.4 Design philosophy

Once the decision is made to adopt an indirect CW system, the remainder of the design concept revolves around the choice of cooling tower arrangement, adequacy of the make-up water source, and safe discharge of the purge water.

4.2.5 Design considerations

Choice of cooling tower

Typically, a large (above 1,000 MWe) power station could be served by four 115 m tall natural draught towers or a single 180 m tall tower. The design issues to be considered in deciding the number of cooling towers and the type of tower include:

- visual impact of towers and plumes;
- noise made by water passing through the towers;
- additional noise made by the plant for mechanical draught towers;
- spray nuisance in the lee of towers;
- fog nuisance produced downwind of mechanical draught towers;
- land requirement;
- construction and operating costs.

Make-up system

The majority of indirect systems take make-up water from rivers and return the purge water to a point downstream of the intake. Hydrological studies need to be carried out to determine the probability distribution for extreme low river flows, taking into account all other uses of the river. An alternative source of cooling water will need to be found if there is a possibility that river abstraction will not be permitted during periods of low flow, such as a nearby flooded gravel pit.

Cooling towers permit a measure of flexibility in the availability of make-up water. Providing the cooling tower ponds have sufficient capacity a power station can operate for an appreciable time without any make-up, for example for the duration of the tidal cycle below mean sea level.

The intake, screening and pumphouse arrangement for a make-up system is similar to that of a direct CW system but on a much smaller scale. For example, the fine screen can be situated at the intake to exclude fish from the system.

The most common problem encountered is siltation. The usual solution is to permit settlement in the tower ponds and to clean out the ponds on a regular basis. In addition, the make-up water can be de-silted by passing it through a settlement tank before it enters the cooling tower circuit. Whichever measures are adopted, the removed silt has to be disposed of in accordance with waste disposal regulations.

Purge system

The design of the purge water outfall is similar to that of a direct CW system but on a much smaller scale.

4.3 Choice of CW System

Past experience in the UK has shown that estuaries or the sea are the only CW source that can support a direct CW system for a large power station. All inland large power stations therefore operate indirect CW systems.

However, greater awareness of the environmental impact of direct CW systems and designation of conservation areas could lead to more use of indirect CW systems in coastal locations. Situations where an indirect CW system may be advantageous include:

- where the environmental impact of the intake and outfall of a direct CW system is not acceptable;
- where the sea can support a direct CW system for one reactor but there is a proposal to develop a power station site with twin reactors;
- where the power station is located on high ground or is several kilometres from suitable abstraction/discharge sites.

A site with twin reactors could have a direct CW system for one reactor and an indirect CW system for the other reactor.

Alternatively, the thermal impact of a direct CW system outfall could be mitigated by a cooling tower in the outfall system (Figure 4-15). Such a tower would be smaller than that for a full indirect CW system because it is only required to reduce the temperature by a few degrees, to comply with the discharge limit. At cooler times of the year it may be possible to bypass the tower and only partial operation of the tower will be necessary.

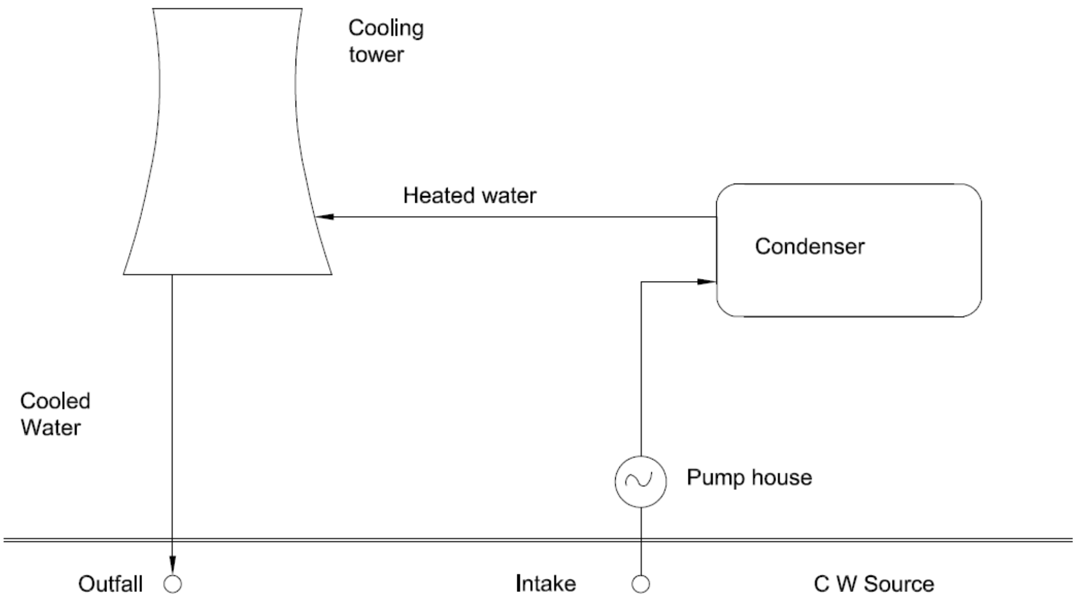


Figure 4-15 Diagram of typical direct CW system with ‘helper’ tower

5 CW system design for new UK nuclear power stations

5.1 Introduction

This chapter discusses the civil engineering design of the CW intakes and outfalls anticipated for the proposed construction of the next generation of nuclear power stations in the UK. The following table gives an indication of the CW demand for the two most likely reactor types:

Table 5-1 Electrical out put and CW demand for the EPR and AP1000 reactors

Reactor	Electrical Power Output	CW Demand (m^3s^{-1})
EPR	1,600	72
AP1000	1,117	57

5.2 Potential sites and water sources

The site of any new power station is ultimately chosen by balancing a wide range of environmental, engineering, technical and commercial considerations. The main civil engineering considerations are:

- ground conditions;
- availability of cooling water;
- proximity to an existing connection to the national grid;
- transport access to the site for the workforce, materials and heavy loads.

Between 1953 and 1963 eleven Magnox stations were built in the UK. The majority of the sites were selected by the then Central Electricity Generating Board (CEGB) after a considerable amount of work (CEGB, 1986). Since 1965, three advanced gas-cooled reactor (AGR) stations and one PWR station have been built alongside four of the Magnox stations (Dungeness, Hinkley Point, Hunterston and Sizewell). In addition, new AGR stations have been built at Hartlepool, Heysham, and Torness. All the current nuclear stations are coastal and each has a direct CW system with a flow in the order of 50 cumecs, somewhat lower than may be expected in the future (Table 5-1).

The proposed third generation of nuclear reactors will have a generation capacity of between 1,200 and 1,600 MWe. The power stations will probably require a direct CW system with a CW flow of between 40 and 80 cumecs per reactor, depending on the temperature rise through the turbine condenser.

It is therefore anticipated that the CW system for each of the proposed new reactors will be one of:

- a direct CW system built at an existing Magnox or AGR site where there is room to expand;
- an indirect CW system built at an existing Magnox or AGR site where there is room to expand but there is a need to mitigate the environmental impact;
- a direct CW system built at a new coastal site;
- an indirect CW system built at a new coastal site where there is a need to mitigate the environmental impact;
- an indirect CW system built at an existing or new inland site with a reliable source of make-up water.

If the CW system option is an indirect system, the make-up water flow will probably be between two and three cumecs.

British Energy has already published Environmental Scoping Reports for four possible new nuclear stations at Hinkley Point, Sizewell, Bradwell and Dungeness[20]. In all cases it is likely that the plant will be directly cooled, with no need for cooling towers.

5.3 Basis of design

This section discusses the civil engineering design of the CW intake and outfall for the direct CW system only. The design of the CW intake and outfall for the indirect CW system is omitted because it would be similar to the design of the many intakes and outfalls that already exist for the major inland power stations in the UK, such as Drax.

The final design of a direct CW system for a reactor will be site-specific because it will depend on unique site characteristics such as ground conditions, topography, bathymetry, tidal range, current regime, ecology and meteorology. It is therefore not practicable to derive specific design recommendations that will be applicable to all possible reactor sites. However, the discussion will cover the general principles of the engineering design and these principles can be transferred to any site.

5.4 CW intake design

5.4.1 Offshore intake

Offshore intake for deep intake conduits

Figure 5-1 shows a typical submerged offshore intake structure for a deep tunnel intake conduit. The structure is a capped radial flow structure, constructed in precast concrete and placed on a prepared granular foundation.

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[20] http://www.british-energy.com/documents/Hinkley_Point_Environmental_Scoping_Report.pdf
http://www.british-energy.com/documents/Sizewell_Environmental_Scoping_Report.pdf
http://www.british-energy.com/documents/Bradwell_Environmental_Scoping_Report.pdf
http://www.british-energy.com/documents/Dungeness_Environmental_Scoping_Report.pdf
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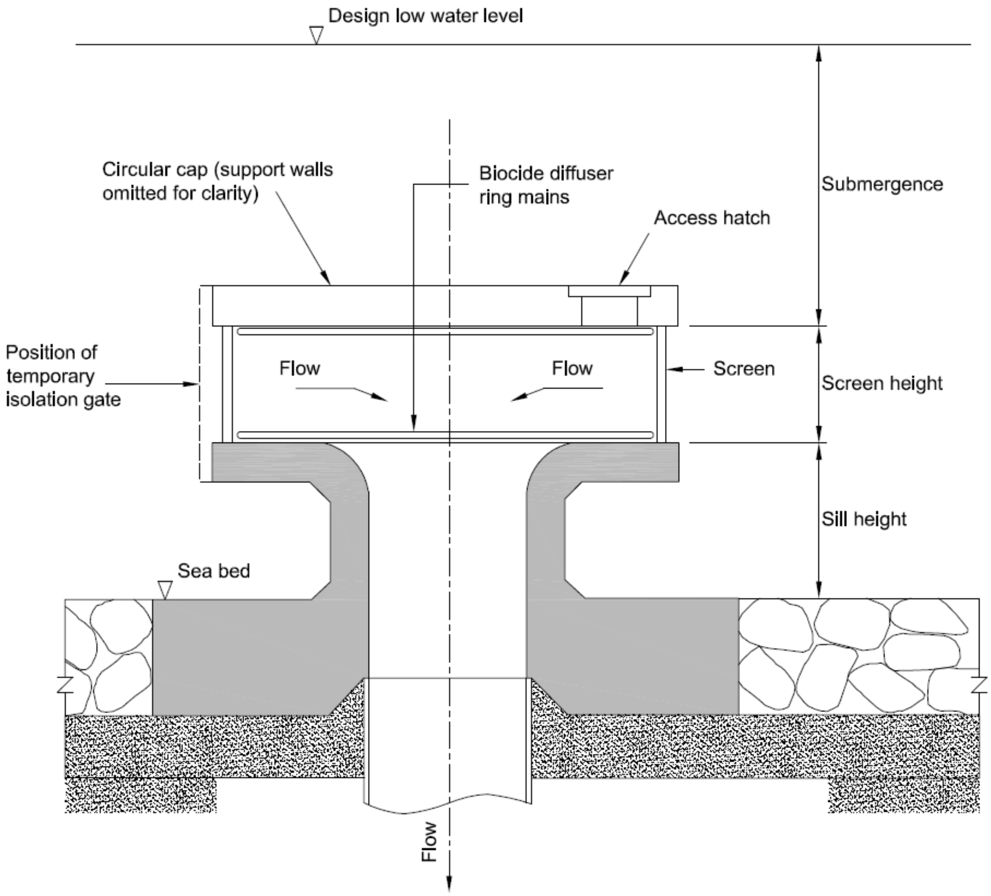

Figure 5-1 Typical submerged intake structure

The design is based on the assumption that tidal currents are small, less than 0.1 ms^{-1}. If tidal currents are such that fish could be carried into a radial flow structure, the hydraulic design of the structure will need to be modified. Upstream and downstream openings of the intake will have to be closed off to ensure intake only draws water in a direction perpendicular to the tidal flow. Such a design exists at present only as a concept that has undergone limited model-testing and will have to be fully evaluated by physical modelling to ensure the resultant velocity through the intake screen will not draw in fish (Section 6.1.6).

The number and diameter of the deep tunnels will probably be decided by the construction contractor. A single tunnel would be adequate for the operation of the power station. Inspection and maintenance of the tunnel could be carried out using underwater engineering techniques during one of the scheduled plant outages (CEGB, 1986). However, access is usually impractical over distances of more than 500 m from either end, unless intermediate entry points are provided, because of limits on diver access and on the use of remotely operated vehicles (ROVs) because of cable drag. The diameter of the vertical shafts needs to be as large as possible for economy and may also be decided by the construction contractor. The number of shafts will be decided by the total design CW flow and the optimum flow per shaft.

It is envisaged that the vertical shaft to the deep tunnel will be drilled from a jack-up barge. The shaft will then be flooded, the section of lining above the seabed removed, and the intake structure placed in position. Temporary isolation gates will be fixed to the intake structure to enable the shaft to be dewatered so that the connection to the deep tunnel can be completed in dry conditions. On completion of the tunnel and shaft works, the intake structure will be commissioned by flooding the offshore system and removing the temporary isolation gates.

Comments on the detailed design of the submerged offshore intake structure are listed in Table 5-2.

Table 5-2 Detailed design of submerged offshore intake structure

Aspect of design	Comment
Design low water	The design low water level adopted for the intake must be one that occurs very infrequently, but there are no hard and fast rules for determining this level. Lowest astronomical tide (LAT) is the predicted tide level that is equalled or exceeded only once in a cycle of 19 years although in practice LAT can be exceeded several times a year through meteorological effects. Invariably a margin of safety is included and the designer must check the impact on intake performance if the chosen DLW level is exceeded.
Submergence	The minimum depth of water above the intake screen should be designed to avoid the following at design low water: air being drawn into the intake by the formation of a surface-piercing vortex; the intake cap being exposed in the trough of a wave; and excessive recirculation caused by drawing in warm water from the outfall plume.
Screen area	The required screen area is governed by the design flow for the vertical shaft and the required uniform velocity through the intake screen that will allow fish to escape.
Screen height	The screen height needs to be as large as possible to minimise the diameter of the intake structure and facilitate access by divers for maintenance and cleaning. However, the height must not be so large that the velocity through the intake screen is not uniform with depth.
Screen diameter	The screen diameter needs to be as large as possible to ensure the velocity through the intake screen is uniform with depth. However, the diameter must also be within the appropriate limit for handling the intake as a prefabricated unit.
Screen construction	The dimensions of the intake screen depend on the type of fish or other biota to be excluded. A typical arrangement would comprise rectangular bars 16 mm wide and 50 mm deep, at 90 mm centres. The material of the bars could be mild steel protected by a suitable corrosion protection system. Alternatively the bars could be constructed in stainless steel or a non-ferrous metal that inhibits marine growth. Consideration should be given to making the screen in removable sections to facilitate maintenance and cleaning.
Screen protection	Consideration should be given to coating the intake screen bars with antifouling paint to prevent biofouling. The coating would need to be replaced approximately every five years.
Sill height	The sill of the intake should be high enough above seabed level to prevent sediment and debris being drawn from the seabed into the intake. This also reduces the risk of drawing in benthic fish.
Structure diameter	The diameter of the structure at seabed level should be as small as possible to minimise the impact of the structure on the

Aspect of design	Comment
	sediment transport regime in the vicinity of the structure.
Foundation	The foundation should be designed for stability under the imposed dead and live loading including the combined hydrodynamic loading from the CW flow, extreme currents and extreme waves. The seabed should be excavated and replaced with suitable rock fill material if the seabed is soft below the foundation formation level.
Scour protection	The sea bed around the intake structure should be reinforced with rock armour to prevent erosion of the bed due to increased turbulence in the vicinity of the intake structure.
Biocide dosing system	A system will be required for delivering the biocide to each intake structure in equal quantities. The pipeline can be internal (incorporated within the deep tunnel) or external (lain on the seabed). The pipeline could be prone to blocking and must therefore be designed for full maintenance and replacement if necessary. The biocide should be discharged from a diffuser designed to release the biocide close to the surfaces of the intake immediately downstream of the intake screen. The biocide cannot be released in front of the screen because of the risk that some of it may not be carried into the intake.
AFD system	To meet Best Practice, the intake should be fitted with an acoustic fish deterrent (AFD) system (Section 6.1.5).
Diver access	A hatch should be provided in the cap of the intake so that divers can gain access to the intake for inspection and maintenance.
Isolation gates	Gate supports and guides should be provided to enable the intake to be isolated and dewatered during construction as well as for possible subsequent maintenance during the life of the power station.
Corrosion protection	The choice of metals should ensure that corrosion cells are not set up which have a harmful effect on the intake structure.
Navigation	The location and extent of the CW intake should be marked with appropriate navigation aids to satisfy the requirements of the local navigation authority. The design of the intake structure should be checked for accidental ship collision.

The minimum depth required for an offshore intake is site-specific. Typically, a minimum depth of 8 to 10 m below LAT could be required. The position of an offshore intake is also site-specific and needs to be decided in conjunction with the outfall position. In addition, the intake must be located in an area where the seabed is open and free from obstructions so the intake can draw in the CW flow without distorting the ambient flow regime significantly.

Offshore intake for shallow intake conduits

If the offshore intake conduits are constructed by the immersed tube technique, the intake structures will probably be constructed as an integral part of the immersed tube units. Comments on the detail design of the intake structure are the same as Table 5-2 with one exception. The need for isolating gates does not apply to shallow offshore intake conduits because they cannot be dewatered owing to flotation.

A key aspect of the immersed tube design is that major temporary works are required in the inter-tidal zone where the offshore construction method changes to the onshore construction method (Figure 5-2). These works can have an environmental impact.

Figure 5-2 Temporary works for immersed tube construction, South Humber

5.4.2 Inshore intake

The detailed engineering design of an inshore intake depends on whether the intake is in a sheltered location (Figure 4-3) or an exposed location (Figure 4-4). Essentially the civil engineering structure will be subjected to coastal processes and will need to be designed to take account of tidal range, current regime, wave action and sediment transport under prevailing and storm conditions.

The design comments in Table 5-2 are also generally applicable to inshore intakes. In particular, the comments with respect to design low water, submergence, screen area, screen construction, screen protection, sill height, scour protection, biocide dosing system, AFD system, corrosion protection and navigation are relevant.

5.5 CW intake screening

5.5.1 Why cooling water needs to be screened

Power stations usually require multiple CW screening systems to prevent the entry of unwanted debris, biological detritus and living organisms into the tunnels and various parts of the plant where they might cause damage or operational difficulties through blockage of flow. Any new directly cooled coastal plant should provide equipment to ensure that the plant can operate at its normal capacity for nearly all of the time.

Screening is required not only to protect the plant but also for public safety (divers, swimmers and anyone falling into the water) and for the exclusion of fish, marine mammals, diving birds and other biota. Protection of aquatic life is a relatively recent factor in intake screening design and was seldom taken into account in CEGB power station designs prior to the 1970s, except in special cases such as Oldbury-on-Severn and Pembroke power stations, where rudimentary fish rescue systems were contrived to assist the return of entrapped salmon and sea trout smolts to the wild[21].

Some sites are particularly prone to invasions of biota, such as sprat shoals or jellyfish, while others may be afflicted by inundations of kelp and other seaweeds. Although careful planning and siting help to minimise these risks, environmental conditions can change and the unexpected can occur. Even without these transient events, everyday loadings of screening systems can be large, representing an environmental nuisance and disposal cost that stations would prefer to avoid.

The main reason for installing screening at any power station is to protect the downstream equipment. The first item is the main CW pump that moves the water through the cooling cycle. Modern concrete volute pumps or vertical spindle mixed flow pumps generally can tolerate particles of 25 mm in diameter without problems. However heat transfer takes place in the condenser, usually constructed with tubes having an internal diameter of between 20 and 25 mm, through which the cooling water flows to cool the steam passing through the condenser on the outside of the tubes. A standard condenser will have many hundreds of tubes and the heat transfer efficiency is vital to the overall performance of the power plant. It is usual practice today to have screen openings one-third of the condenser tube internal diameter, that is 6-8 mm. In almost all of the screens in current UK power stations, nuclear or conventional, the mesh installed on the band screen or the drum screen is between 5 and 10 mm.

An important factor influencing the size of screen mesh is hydraulic head loss: the smaller the mesh, the greater the head loss. With the large flows of water involved, the pumping energy must be taken into account. Head loss increases as a square of the velocity and the smaller the mesh the greater the head loss will be, as velocity of water passing through the mesh for a given flow will increase with reduction in the open area of finer meshes. Increasing the size of the screening plant can offset this. The size of the screening plant is therefore selected at the design stage and later reductions in mesh size may be difficult.

Finer meshes also remove more debris, and are more prone to blockage, a further factor that needs to be taken into account.

In other processes within the cooling plant, a much smaller mesh size may be needed to protect, for example, electro-chlorination equipment or screen wash water jets. In these cases, as the water quantities are small, separate automatic pipeline strainers are employed and these can polish the water to the required degree, usually to 0.5-1.5 mm.

5.5.2 Stages of screening

Modern CW intake designs therefore incorporate up to four levels of water screening:

- coarse or fine screening at the primary intake point;
- onshore coarse screening, ahead of fine (drum or band)screens;
- main plant fine screening by band or drum screens;
- auxiliary cooling water supply filtration.

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[21] Unpublished CEGB report *Smolt recovery system, results of 1970 run, Oldbury-on-Severn Power Station*, M-OLD-CWE-1110

The requirement for each of these stages depends on preferences of the designer, as well as local site-specific factors such as debris concentrations in the water.

5.5.3 Screens located at intake point

Coarse screens with bar spacings of between 50 and 250 mm (typically around 90 mm) are normally used at this stage to prevent entry of large items such as plastic drums or tree branches as well as marine mammals, diving birds and divers or swimmers.

Coarse screens of heavy-duty treated steel bar or stainless steel construction are on most modern stations aligned vertically across the intake ports or around the periphery of capped offshore structures. Some older CEGB stations (such as Sizewell A, Dungeness A) which had no cap on the intake used a cylindrical bar cage surmounted by a conical array of radial bars ('Chinese hat' type). This arrangement is now not used since it allows water to be drawn vertically, increasing risk of surface vortex formation (a danger to craft and swimmers) and of abstraction of warmer surface layers. At Hinkley Point A & B station, which share a common intake caisson located 500 m offshore into the Bristol Channel, water enters via coarse screens fitted to both the sides and the top of the structure. In none of the offshore structures is any automated raking mechanism provided, owing to the hostile sea conditions to which these structures can be exposed. The screens do, however, biofoul and clog with rope, fishing nets and other debris with time and these accretions have to be removed by divers during planned outages.

Where the intake is located offshore, some station designs (such as Sizewell A, Dungeness A, Hinkley Point A & B) have provided a superstructure over the intake. The main purpose has been to support a crane to allow fitting and removal of an intake plug for dewatering purposes. In practice it was found that the structures were difficult to access safely from the sea and became a health and safety risk and a maintenance liability. Those at Sizewell A and Dungeness A were therefore removed during the 1980s. The exception is at Hinkley Point, where an extra culvert was installed as a pedestrian tunnel, which provides access from the shoreline. Turnpenny (1988a) points out that excessive structure around an intake inadvertently creates an artificial reef, increasing the risk of attracting fish into the intake. Nowadays, superstructure is normally confined to that required to support navigation lights and any fish protection systems (see Section 6.1.5).

Wharfside intakes built at the shoreline are normally fitted with vertical bar screens set into the intake openings. It is unusual to provide raking mechanisms but since the locations are more sheltered it is usually feasible to do so in locations where the trash loading is high. For example, Keadby CCGT Power Station on the Trent estuary at times draws in large quantities of brushwood and dead straw-like vegetation and has been fitted with an automated raking machine to alleviate this problem (Figure 5-3). Keeping screens clear of blockage is important, as stations may otherwise breach abstraction licence conditions relating to maximum allowable velocities for fish protection.

Figure 5-3 CW intake at Keadby, showing overhead gantry rail for screen raking system and accumulated brushwood

Some smaller tower-cooled CCGT stations use passive fine screens at the primary intake point. UK examples include the Deeside, Connahs Quay (both Welsh Dee) and Killingholme (Humber) stations, all located on estuaries. These are known as passive wedge-wire cylinder (PWWC) screens, and the screening elements are formed by triangular-section wires wrapped around a cylindrical former (Figure 5-4). The wire spacing is generally specified as 3 mm; this and the design through-slot velocity of under 0.15 ms^{-1} prevent entry of all but the smallest fish larval stages. The screens work best when located in a sweeping flow that exceeds 0.3 ms^{-1} for a high proportion of the time, as this carries away debris and assists in fish escape (Turnpenny and O'Keeffe, 2005). In 2005, when Jacobs reviewed operating experience of PWWC screens, the Connahs Quay station reported good operational performance in the eight years or so since its construction, while at Deeside intermittent blockage problems were reported. At Connahs Quay, the screens are placed well out into the estuary channel, encouraging debris clearance, while at Deeside the screens are recessed into the bank and out of the channel flow.

PWWC screens remain an unproven technology anywhere in the world in marine waters as hostile as those offshore in the UK (eg significant tides, waves and currents etc). The large CW requirements, and consequently huge screen array sizes, combined with the risk of catastrophic blockage, for example by a sudden influx of seaweed during a storm, has so far been thought to make their use unsuitable for nuclear applications.

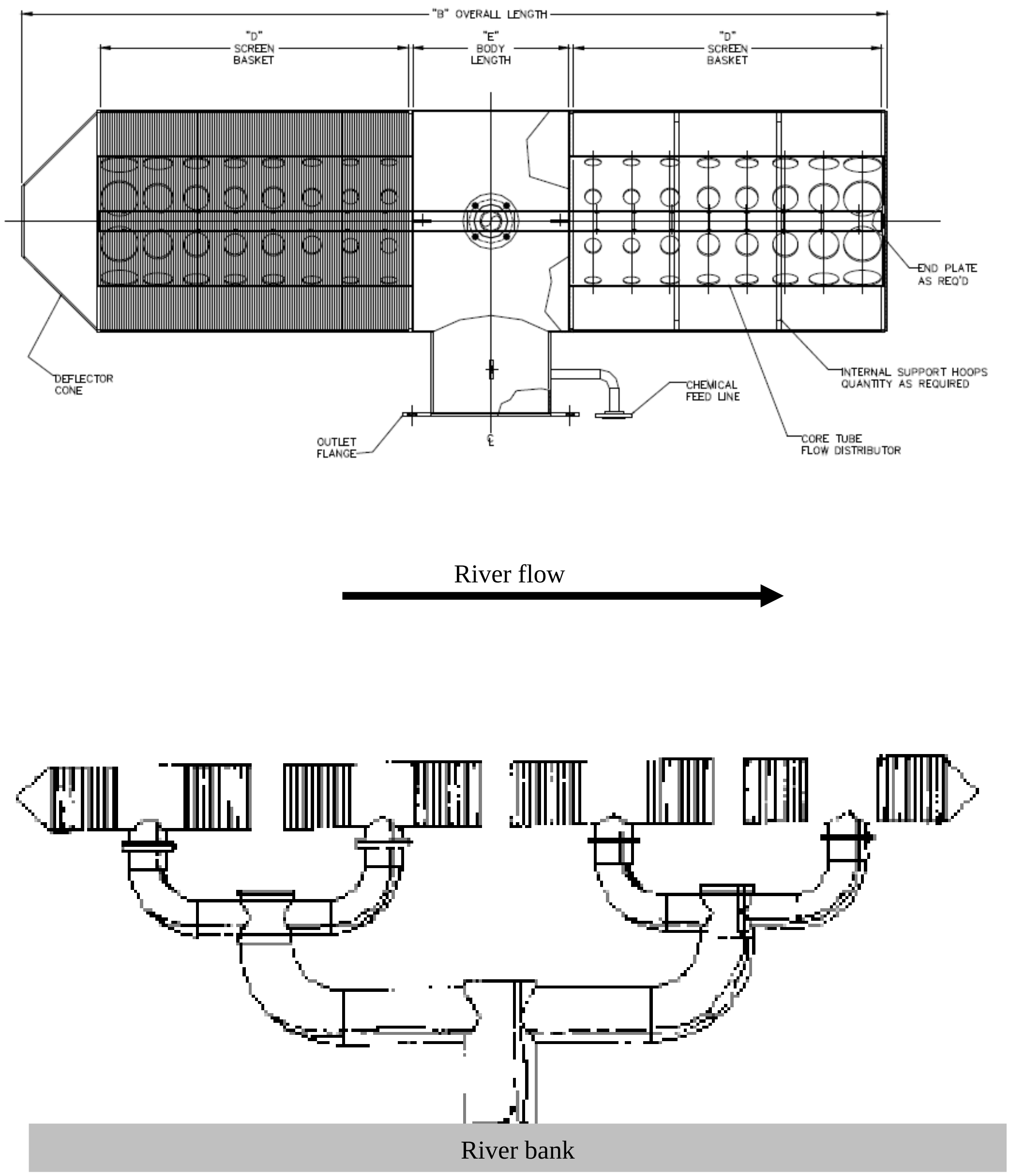


Figure 5-4 Top: passive wedge-wire cylinder screen in 'T'-format. An internal core-tube with holes of varying diameters is used to obtain an even flow distribution across the screen surface (courtesy Hendrick Screens). Bottom: PWWC screen array for bankside use.

5.5.4 Forebay bar screens

While not found on British nuclear sites of the CEGB era, it is now common for directly cooled power stations to install automatically raked bar screens into the forebay ahead of the drum or band screens. Such bar screens typically have a clear spacing of 50 mm. The raking mechanism is similar to that shown Figure 5-3, and one rake normally traverses all the screen sections either at preset time intervals or when a head differential is detected across the screen. In the UK, these intermediate bar screens are to be found, for example, at Great Yarmouth and South Humber Bank, both directly estuary-cooled CCGT stations. In continental Europe they are also used at the Doel nuclear plant on the Scheldt Estuary in Belgium.

The use of forebay raked screens is becoming more common and they may well be used to reduce fine-screen loading on new nuclear stations, especially where drum or band screens are fitted with a mesh finer than the 8-10 mm common on CEGB-era stations. An important consideration will then be their ability to handle larger fish safely. Fish recovery and return (FRR) systems presently in use rely on the band or drum screens to handle the fish and, to date, little or no thought has been given to damage that may be incurred at this stage. First, it should be recognised that larger fish may become pinned to the screens for long periods between raking operations; secondly, the lifting mechanisms will be unlikely to retain larger fish such as eels and adult lampreys, which may as a consequence be subject to multiple handling by the rakes before successful removal.

Observation of the operation of raking systems has revealed that problems can occur when the flow conditions into which they are installed are not fully understood. It is not uncommon to see debris that has been brought up by the rake to the top of the screen being washed off again before exiting the water by upwelling currents in front of the screen. Regulators should be satisfied that the design and performance of any forebay raking system is compatible with FRR requirements (see Section 6.1.6).

5.5.5 Fine screening

Travelling band screens

Travelling band screens (or just ‘band screens’) comprise articulated bands of mesh panels attached to chains driven by sprockets (Figure 5-5). The band of moving mesh is introduced in front of the pumps, the screens elevating debris from the water to deck level where it is washed off by water jets. As more sophisticated travelling band screens have been developed, they have changed from straight-through type to dual flow and centre flow, referring to the flow pattern of water passing through the screen:

- Straight through – All of the water passes from one side to the other through both the ascending and descending sides of the moving band.
- Dual flow – Mesh panels move parallel to the flow, half passes from outside of the centre through the ascending side of the band, half passes from the outside to the centre through the descending side of the band, all of the water passes out through a single exit often referred to as the back opening. This type of screen is used in the UK and France.
- Centre flow – This machine also has panels positioned parallel to the flow but in this case all of the water passes into the centre of the machine with

half passing out through the descending side and half through the ascending side. This type of screen originated in Germany and is still the German standard.

These flow patterns are shown below in Figure 5-6.a, b and c.

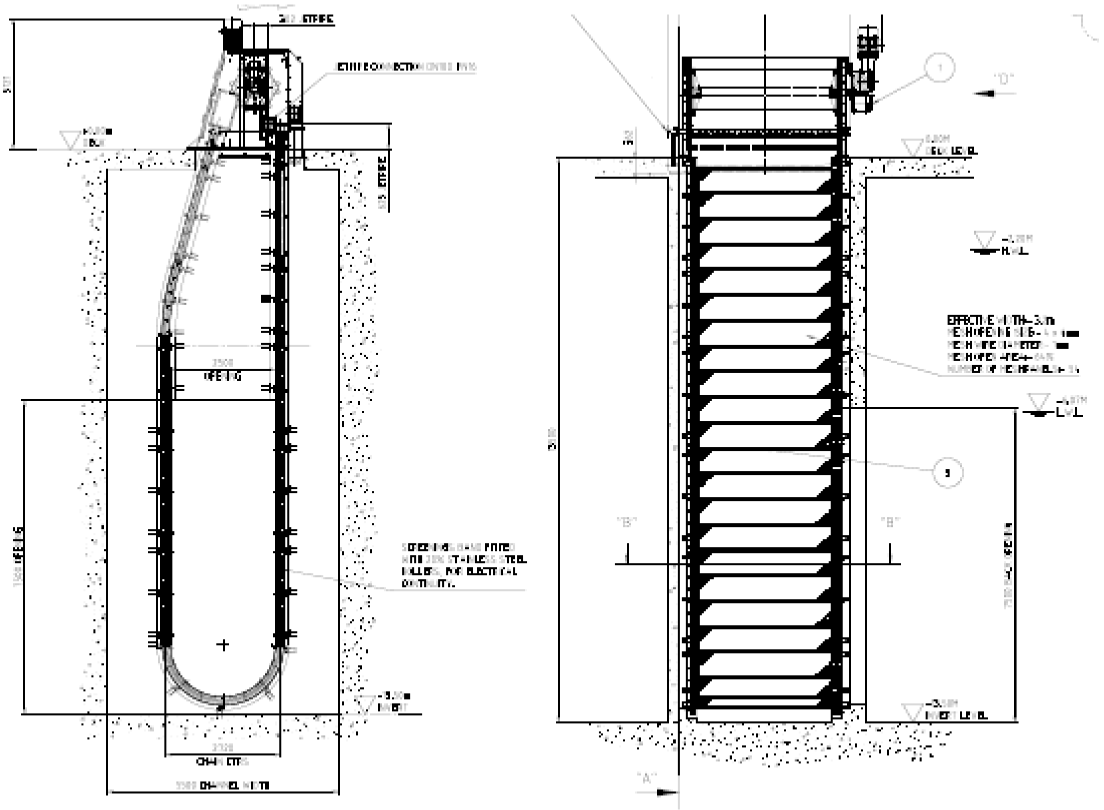
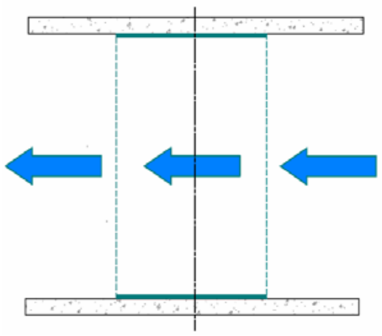


Figure 5-5 Engineering drawings of travelling band screen: vertical section (left) and elevation (right)

(a)

Bandscreens: Flow Pattern

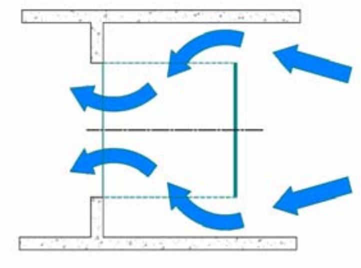


Through Flow (TF):

- Can “carry over”.
- Produces a parallel flow.
- Simpler civil work requirement than for DF or CF types.
- Flow has to pass through the mesh twice, can lead to higher head losses with fine mesh.
- Predominately used in USA.

(b)

Bandscreens: Flow Pattern

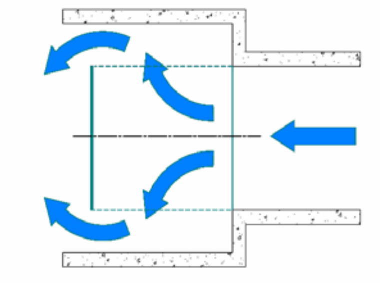


Dual Flow (DF):

- No "carry over".
- Produces a converging flow, therefore suitable for close coupling to CW pump.

(c)

Bandscreens: Flow Pattern



Central Flow (CF):

- No "carry over".
- Produces a diverging flow, therefore not suitable for close coupling to CW pump.
- Good for heavy/large solids quantities, therefore commonly used on waste water applications.

Figure 5-6 Characteristics and flow arrangements for travelling band screens in plan view: a) through-flow, b) dual flow (UK and France), c) central flow (Germany)

In the late 1920s there were parallel developments in the UK and France of an alternative type of screen. The need for this screen was driven by a number of factors.

- The moving band of mesh panels on a band screen are carried on a chain comprising links, pins, bushes and rollers. A single machine can have 2,000 moving parts, all of them operating in sea or river water containing sand or silt. As there are often several units on a single plant, the spare part and labour costs are a significant factor in plant running costs.
- A modern power plant or industrial complex may need 50 to 100 m^3s^{-1} CW, whereas a single band screen has an upper limit of 10 m^3s^{-1} capacity, governed by the structural integrity of the machine. With the need for standby machines, high capital and running costs become a major factor in screen type selection.

These constraints led to development of the rotating drum screen. The 1930s saw an increase in the use of drum screens in France and the UK. Curiously, the drum screen concept never evolved in Germany or America. In France, development was led by Beaudrey and in the UK by F.W. Brackett (now Eimco Water Technologies). These two variants will be referred to here as the French and British designs, respectively.

Owing to the large CW requirement of nuclear plants, drum screens are much more likely to be used in UK new nuclear build for fine screening than band screens. One important aspect of this outcome concerns fish recovery and return methods, which, at present, are simpler to install and therefore further advanced on band screens than on drum screen systems.

Drum screens

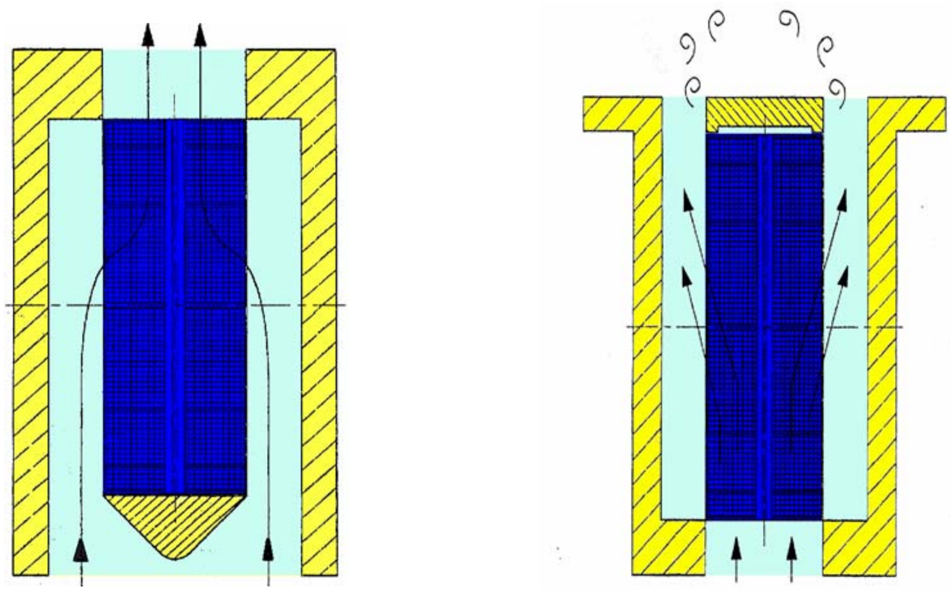
Drum screens are much simpler in construction than band screens, comprising a slowly revolving mesh cylinder through which the water is passed. This essentially has only one moving part, the screen itself, although a drive motor and washwater pumps are required as well.

In the UK the drum screen evolved with a water flow pattern of ‘in-to-out’, water passing into the middle of the screen from both sides (in the case of double-entry drum screens) and outwards through the mesh, catching and arresting all the debris on the inside of the screen (Figure 5-7a).

In France the drum screen evolved using a flow patter of ‘out-to-in’ water passing from the outside of the drum through to the centre of the drum, retaining debris on the outside of the drum (Figure 5-7b).

There are variations of orientation in both types and when water passes into the screen from one side only (in the case of ‘in-to-out’ screens) these are designated single entry drum screens (SE); the ‘out-to-in’ flow pattern can also be arranged with the main shaft parallel to or at right angles to the flow. These variations are often related to site constraints or pump station layouts.

(a) In-to-out **(b) Out-to-in**



The in-to-out flow pattern produces a converging flow. This minimises the outlet turbulence and distance required between the screen outlet and CW pump inlet.

The out-to-in flow pattern produces a diverging flow that can result in turbulence at the outlet and requires the flow to be converged to meet the CW pump inlet, increasing the distance between screen and CW pump.

Figure 5-7 Schematic showing in-to-out (UK) versus out-to-in (France) drum screen design concepts

The upper limit of a single drum screen is typically around 35 m^3s^{-1}, well above band screen capacity. Multiple screen installations with four per site are common on UK and French sites. Of these, three may be adequate to handle the full CW flow but a fourth provides a standby and additional capacity that can be brought online as required. Nuclear plants often operate all four screens continuously.

Interestingly, the French steadfastly have stood by their 'out-to-in' concept and the British, their 'in-to-out' concept. The following is a summary of the statements that British and French screen suppliers regularly make in putting their cases forward.

For the French system, the manufacturers state the benefits of their out-to-in concept as follows:

- Debris remains on the outside of the drum screen and is visible to the operator.
- Spray water jets (by virtue of the design) are on the inside of the screen and are easily accessible.
- The drum screen floats if the mesh becomes blocked to the point where holding down bolts snap.
- Gravity is of no assistance in removing debris from the screen and only the washwater pressure is significant.
- Due to the tendency of the screen to float, bearing loads and drive power requirements are reduced.
- Debris cannot fall off the screen.
- The trash collection trough is continuous over the full width of the screen.
- The depth of the screen sump can be reduced with out-to-in screens.
- Screens with out-to-in flows can be constructed to larger sizes.

For the British system, the manufacturers state their arguments promoting the in-to-out flow as follows:

- Washwater jets are vital to the operation of the screen; these are external to the screen and easily accessible without stopping the screen.
- Debris is removed from the screen by a combination of gravity and low pressure washwater.
- Hydraulic loads are absorbed by the concrete structure.
- Drive gear is external and therefore always accessible.
- Roller bearings are designed to last the life of the screen.
- The screen is hydraulically balanced.
- Flow to the pump is laminar and not turbulent.

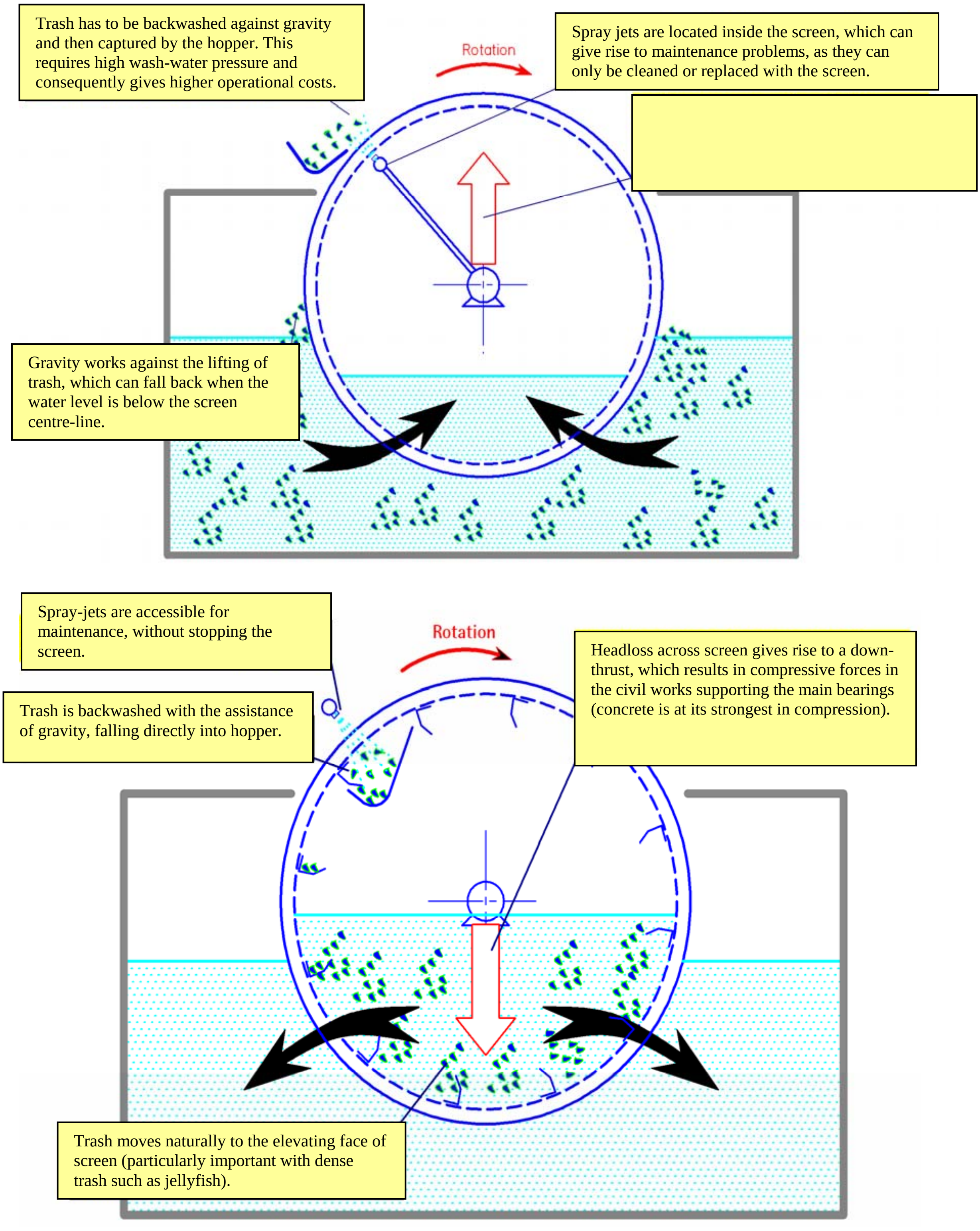


Figure 5-8 Further comparison of out-to-in and in-to-out drum screen designs

A drum screen of either configuration will only continue to operate if the screen presents a clean mesh surface to the water being screened. Washwater jets clean the mesh by sending a fan of pressurised water to remove debris from the screen. Jets do unfortunately occasionally block. Jets on the in-to-out type of screen are accessible from the operator's platform without stopping the screen. On the out-to-in, screen jets are inside the screen, the screen has to be stopped before a jet can be cleaned. British screens require 1.7 Bar pressure to clean the mesh while French screens require 3.0 Bar. Low pressure sprays are beneficial for safe fish recovery (see Section 6).

British screens elevate the debris to the rubbish hopper with internal elevating trays, French screens with external elevating trays. The rubbish hopper on the British screen is positioned about 30º from vertical and is thus positioned to ensure that elevators tip their contents into the rubbish hoppers. During the 1.5 to two metres of travel over the rubbish hopper, low pressure water flushes debris (and live fish) from the mesh with the aid of gravity. With the French design, debris has to be blown off the mesh against gravity (Figure 5-8), a system that would more likely damage live fish as they have to be knocked off the mesh into the external rubbish hopper. This may be of concern when considering adaptation of the screens for FRR (Section 6.1.6).

Use of band and drum screens in other countries

The advantages of drum screens have been recognised in almost every country where large sea, river and estuary intakes are used. Manufacturers' supply tables (Beaudray, Eimco Water Technologies) give an indication of their geographic use. The French manufacturers have supplied around 500 drum screens; of these, 250 are in France, 60 are in Holland, 27 are in Belgium, others in various parts of the world. Eimco (formally Brackett) has supplied around 1,100 drum screens, of which 71 per cent were exported from the UK.

China, which is a rapidly growing market for nuclear and coal-fired plants, has adopted the in-to-out design. Hong Kong has virtually always built plants with the in-to-out flow pattern and following its reintroduction to China, has continued this trend.

China tried the out-to-in screens in 1987; six were installed at the Daya Bay plant in Guangdong. Following that, four in-to-out screens were purchased for Ling Ao 1 and after two years of operation four more were ordered for Ling Ao II. Quinshan and Dalian followed with in-out designs: a total of 20 drum screens. All the new plants planned in China are intending to use in-to-out drums.

However, of two nuclear power plants using EPR reactors being built in Europe one, Olikiluto in Finland, has selected band screens. The other, at Flammanville in France, has selected out-to-in drum screens.

Around the world, there are many hundreds of screens installed at sites with flow configurations out-to-in, in-to-out and with dual and single exits. It is important that pump suppliers have no issues with the type of screen that is protecting their pumps as the screen can often affect pump performance and, beyond the pumps, downstream equipment such as condensers, plate heat exchangers or other process equipment.

On large projects a model test is often carried out and can reveal performance problems or hydraulic concerns. Testing facilities are found at organizations such as HR Wallingford or Hydrotech in the UK or Alstom at Nantes in France.

In conclusion, the out-to-in flow pattern is predominantly used in France and in-to-out flow pattern is the established norm in the rest of the world. However, Beaudrey are able to build in-to-out screens and there are around 20-25 units in their reference list of in-to-out configuration. In recently built and upcoming nuclear power plants to be built by French contractors or developers outside of France, the in-to-out flow pattern has been selected. Examples are: (Alstom) Nuclear 1 South Africa, (Alstom) San-Men China, (CGNPC) Ling Ao1 and Ling Ao 2.

Figure 5-9 Overhead view of typical set of four drum screens at a coastal plant

Pressure strainers

In the majority of CW intakes the screening that takes place prior to the pumps uses a gravity system, where the intake is open to the atmosphere and water passes through the filters from the pump drawing water through the screens.

In the early 1960s, CEGB were considering all available options for CW screening and there was a move to use a pressure filter as the primary filter on the downstream side of the main CW pumps. This was a challenge as the quantity of water being pumped involved large diameter pipework and velocities in excess of those normally considered for automatic filters. The challenge was taken up by one supplier that based its design on a smaller scale pressure filter supplied to an industrial plant in Deptford UK. The new design of the scaled-up plant was approved by the CEGB and was selected as the main intake filter for the nuclear plant constructed at Bradwell in Essex. This design was known as the Deptford strainer and consisted of an internal rotating filter element that was backflushed using a portion of CW flow at the discharge pressure generated by the main CW pumps. This bank of filters, one per pump, was commissioned in 1962. Deptford strainers were also used at Aberthaw and Isle-of-Grain power stations.

The design was reasonably successful but proved expensive to maintain and the filters suffered regular blockages throughout the life of the plant. This type of screening soon fell out of favour and from about 1970 new plants reverted to open gravity systems using band or drum screens.

Automatic pipeline filters do have their place in cooling water systems but not as the main filter for the cooling water system. They are essential as fine filters for the various downstream processes but have a limited capacity, which in the extreme is in the order of 2 m^3s^{-1} and in most cases is around 0.5 m^3s^{-1}.

5.5.6 Trash handling and disposal

Both band and drum screens are designed to backwash any filtered debris and biota into troughs, known as ‘launders’, which convey the material into a collecting basket or some other receiver. Washwater pumps, usually abstracting from the screenwell, supply both the spray jets that clean the screens and the water to flush debris along the troughs. Flow rates are usually quite low, in the order of 4 ls^{-1} per screen and pressures may range from between one and four bar, depending on manufacturer (see above discussion of drum screens).

At most CEGB sites, operating licences required backwashed debris to be drained and transported off site to landfill. This remains the case at most sites today. In a few cases, debris is put back to sea via the thermal discharge culvert, either direct from the screens or after maceration.

In all cases, stations require consents for disposal back to water. This is now commonplace where fish recovery and return (Section 6.1.6) is operated.

5.6 Biofouling control

Main cooling and auxiliary circuits

The extensive submerged surfaces of cooling water (CW) circuits offer ideal conditions for the growth of bacterial and fungal slimes (micro-fouling) and of large sessile organisms such as mussels, barnacles and hydroids (macro-fouling). These organisms arrive mainly as spores and larvae that pass easily through the screens protecting the intakes. Once settled, the steady flow conditions provide an abundance of food whilst hindering the establishment of motile predators. Macro-fouling can restrict flow and cause blockages and leaks in condensers and heat exchangers, leading to reduced efficiency and plant outages. Even a thin film of slime can significantly reduce heat transfer across heat-exchange surfaces, whilst beneath it corrosion may be accelerated. The lost output from these causes would have to be replaced from potentially older, less-efficient stations.

Many approaches have been tried to combat biofouling but the majority, both chemical and non-chemical, have proved to be ineffectual and most operators favour continuous low-level chlorination (Turnpenny & Coughlan, 1992; Jenner *et al.* 1998; BREF, 2000). Chlorination has the advantage of a long track record of human and environmental exposure, unmatched by any of the alternatives. Some, such as ozone, bromine chloride, and hydrogen peroxide, produce essentially the same byproducts as does chlorination so, despite their higher cost, there is no benefit to the marine environment. Although chlorine gas is a powerful respiratory poison, in dilute solution it is an effective, broad spectrum disinfectant. The level used for marine biofouling control (0.2-0.5 mgl^{-1}) is significantly less than that found in most drinking water (5.0 mgl^{-1}).

The use of chlorine gas at UK power stations was discontinued many years ago and was replaced by electrochlorination and sodium hypochlorite (a chlorine source best known as household bleach). The biocidal activity of chlorine solutions is related to their ability to oxidise and they are often referred to as “oxidant”. The main period of chlorination will be when sea temperatures exceed 10°C, typically from May to November. Some oxidant inevitably will be discharged with the cooling water. This constitutes a waste, as well as a potential threat to the environment, so the dosing will be carefully controlled. Sometimes it is feasible to chlorinate only part of the CW flow at any one time and the recombination of treated and untreated streams will reduce oxidant concentrations prior to discharge. This is not possible with a single CW intake and tunnel. The discharge concentration is regulated by the Environment Agency; consent levels have been progressively reducing to a common 0.2 mgl^{-1} (200 μgl^{-1}) and there is now a proposed interim UK EQS of 10 μgl^{-1}, possibly decreasing to 1.0 μgl^{-1}. After discharge, concentrations continue to diminish through further dilution, additional demand introduced by the receiving water and by continuing decay reactions.

Electrochlorination results in the release to atmosphere of hydrogen gas (one molecule of hydrogen for each molecule of chlorine), which represents a small but possible explosion risk on site. The seawater leaving the electrolyser cells usually enters a tall, open-topped cylindrical "detraining" tank from which hydrogen is released to the atmosphere. An alternative method (not from one of the market leaders) uses a mechanical ventilation system that abstracts hydrogen from the top of the electrolysers. The detraining tanks or ventilation stacks have to be sited clear of other buildings and at some sites (such as China's Pearl River delta where thunderstorms are frequent) they are surrounded by lightning conductors. A combination of muco-polysaccharides and outgassing hydrogen in the detraining tanks can result in the formation of lighter-than-air foam (hydrogen surrounded by sodium hypochlorite) that can be blown around the site. This poses a risk to eyes and skin, to metal and concrete structures and, as the foam settles to the ground once sufficient hydrogen has diffused out of it, substantial drifts may accumulate in sheltered corners. These drifts are an explosion hazard. Foam formation can be minimised by water sprinklers at the top of the detrainment tanks.

Inlet works surface coatings

Chlorine and other soluble biofouling control chemicals may not be viable upstream of screening systems where fish recovery and return (FRR) is practiced, owing to potential toxicity to fish (see Section 6.1.6 below). Without treatment, there is a risk at some locations that the surfaces of inlet tunnels, forebays and screenwells across which seawater is flowing will develop fouling. This can impede flow and may slough off and block screens. Where spare tunnels and so on are provided, they may be taken offline and treated periodically by shock dosing with a soluble biocide. Alternatively, various kinds of biofouling-resistant coatings can be used to treat the surfaces at risk. At present, this is a relatively new issue that has arisen from best practice advice (Turnpenny and O'Keeffe, 2005) and some guidance may therefore be helpful.

Non-toxic coatings

Non-toxic coatings work by creating non-stick surfaces on which fouling cannot settle or, if settled, can be pulled off by currents - hence the alternative name of foul-release coatings. There are two categories of non-toxic surface: ablating hydrophilic polymer films and low free surface-energy polymer films. The antifouling properties of hydrophilic polymer films (such as sulphonic acid based copolymers) stem from their self-polishing/self-smoothing ability. To be effective this needs a flow of nearly 10 ms^{-1} and so is of little use in CW systems, where flows are under 3.0 ms^{-1}. The efficiency of these low free surface-energy films (analogous to Teflon coatings) is less susceptible to flow rate. Silicone-based coatings must be applied to a clean, dry surface (less than five per cent moisture for concrete) so it would be difficult or impossible to coat most existing structures. Two samples inhibited fouling in intake bays for five years: *Bioclean* (CMP Chugoku) and Kansai Paint's *Biox,* but most data indicate a far shorter period of efficacy. In 1992 30 per cent of Japanese marine power stations were using silicone-based compounds and most were being repainted every two years. Some had a four-year interval but this does not mean that the coatings were still efficient, although in trials in the US and Denmark silicone coatings continued to be protective in the fourth year after application. Most brands require two or three base coats, applied in low-humidity conditions, although experiments using a single coat have also shown good results. In most trials fouling was still able to settle and grow, but could be dislodged easily by brushing or jetting. The use of a non-toxic coating in screenwells of new nuclear stations would confer the dual advantage of reducing the mass of fouling to be removed and making it much easier to dislodge. However, silicones have a physically soft surface, susceptible to impact and abrasion damage. Attempts to toughen them have always resulted in diminished antifouling performance since most appear to rely on silicone oil exuding from the silicone elastomer (matrix). Looking to the future it is envisaged that the natural antifoulants used by sessile marine organisms to prevent other organisms from growing on them may be used. The structures of several such compounds have been elucidated and a few have been tested under field conditions. However, even when a suitable compound is found there will still be the hurdles of production and application to overcome.

Toxic coatings

Toxic coatings are better known, and are usually in the form of antifouling compounds (paints) or laminates that incorporate bioactive material such as inorganic zinc and copper (oxide or metal powder). The more effective but environmentally damaging tributyltin oxide (TBTO) is no longer available.

Antifouling paints were first used in UK power stations in the 1950s. The antifouling properties of these early paints rapidly decreased (in less than a year) as the surface layer became depleted of copper. Newer formulations have ablative and self-polishing properties by which the toxic surface is constantly replenished as the paint matrix wears away. However the flow rate in culverts does not match that required for optimum ablation and polishing. Modern antifouling paints often are boosted by the addition of biodegradable organocompounds to the mix, although recently simple metal coatings have regained favour. Copper-epoxy paint is essentially metallic copper powder suspended in an epoxy matrix: after application the paint surface is “activated” mechanically, by brushing or sanding, to expose the copper particles. The surface is hard and neither ablates nor leaches copper at a significant rate. These paints are more expensive than the ablative paints.

Copper sheathing

One of the earliest recorded methods of protecting boats from fouling and boring organisms was by sheathing the (wooden) hulls with thin copper sheet. In recent years the availability, and reduced cost, of the more durable cupro-nickel alloys have revived interest in this approach. The basis for the antifouling properties of cupro-nickel is not clear. It has a lower corrosion rate than copper, so it cannot be due solely to the rate of leaching of copper ions from its surface. The antifouling properties of both copper and cupro-nickel appear to stem more from hydrolysis yielding an unstable cuprous hydrochloride film that sloughs off in moving water, rather than to inherent toxicity. Small craft can be constructed of cupro-nickel whilst a larger vessel can have thin cupro-nickel sheet bonded to the hull. Such sheets were bonded to the intake headworks at Seabrook Power Plant (New Hampshire, USA) and apparently gave good fouling protection although there were some problems with bonding the sheets securely. The presence nearby of mild steel components has been reported to suppress the formation of the hydrochloride film. There are also (unconfirmed) reports that reinforcing steel in the underlying concrete can suffer if it has not been plated with cupro-nickel. However, there are many examples of where the use of cupro-nickel sheathing has protected the normally heavily-fouled and corroded inter-tidal zone on the steel legs of oil platforms. For best results, the cupro-nickel is electrically insulated from the underlying steel.

Sheathing offers a long-term solution, but the initial cost is high. Cupro-nickel mesh (expanded metal sheet and woven wire) reduces the cost but sacrifices durability. In the 1970s neoprene (polychloroprene) rubber sheet impregnated with TBTO was used as sheathing (Goodrich "No-foul") but did not gain wide acceptance and fell into disuse once environmental problems with TBTO were recognized. The technology was revived in the 1990s, using 3-mm thick sheets of polychloroprene impregnated with 90-10 cupro-nickel rods (1-mm diam, 1-mm long) or mesh. About 30 per cent of the surface was metal which remained firmly in place, despite the particles losing weight through corrosion, because the rubber matrix tends to expand slightly with age and submersion. The same trial also used an adhesive-backed 90-10 cupro-nickel foil. All products restricted the development of macrofouling and, when present, it could be wiped off fairly easily. The foil thinned at an average 5.5 μm per annum during the seven-year trial. Some reduction in adhesive bond strength was noted, being less pronounced on steel than on GRP (Campbell, Fletcher and Powell, 2004). As with any type of sheathing, the strength of the adhesive bond is a major concern.

The foil system was marketed for a time by EcoSea (Southampton) as Cuproguard™ but it has since been discontinued in favour of a cupro-nickel coating system (paint) Cuprotect™. This has minute (50 to 100 μm) cupro-nickel spheres held in the surface layer of a multi-layer resin coat. The product is claimed to have a 20-year service life.

5.7 Essential cooling water supply

The term 'essential cooling water' (ECW) refers to provision of back-up supplies for emergency cooling of the reactors in the event of a problem. It is therefore a nuclear-specific requirement and can be dealt with in a number of ways.

The abstraction rate of a mid-sized nuclear power station using a direct cooled CW system will be in the order of 50 m^3s^{-1}. The history of existing plants in the UK, Europe and elsewhere shows that from time to time abnormal events occur, and these need to be considered when deciding on the way in which ECW is provided. These events could include screen blockages, earthquakes, tsunamis, once-in-a-thousand-year high tides or manmade events such as terrorist attacks.

In some designs a completely separate cooling water system is provided, built to standards that enable the equipment to withstand the events mentioned above. In other cases the main plant is sized extremely conservatively and provided with duplicate or triplicate drive systems that virtually guarantee the provision of the ECW.

The two EPR reactor plants under construction in Finland and France have adopted different approaches, the one in France adopting a separate ECW screening system and the one in Finland making provision of the ECW possible through special designs built into the main CW system.

Power plants in Europe do get closed down due to cooling water intake problems. In the UK, as previously mentioned, this has involved massive sprat influxes that have closed down or required drastic load reductions at Sizewell A, Dungeness A & B and Peterhead. Inundations of kelp and other seaweeds have similarly caused problems at Torness and Hinkley Point.

If France, the small 'rose-de-mer' jellyfish (known here as ctenophores or 'sea gooseberries') closed the Paluel nuclear plant, and in Sweden, the USA, Far East and Middle East, plants have been closed down with massive influxes of larger umbrella-type jellyfish. In Britain, the only plant that has suffered jellyfish problems is Hunterston, where in 1991, the AGR reactor was shut down by a massive jellyfish influx (Nuclear

News, October 1991). Jellyfish are abundant in British coastal waters through the summer months and warmer conditions (often attributed to 'global warming') increase abundance and exacerbate this risk.

One factor that appears to have protected most UK stations from CW screen blockages by ctenophores is the larger band or drum screen mesh size used. Ctenophores are slightly larger in diameter than the 8-10 mm screens used on most CEGB-built stations but will readily distort under pressure and squeeze through the meshes, being further broken down during passage through the pumps and condensers. While the release of protein can cause foaming at the outfall, ctenophores have not caused the plants to shut down. Conversely, French plant screens are usually fitted with fine meshes of around 2-3 mm aperture. Thus, choice of mesh size is an important consideration in balancing the need for security of supply and safe fish handling (see Section 6.1.6).

5.8 CW outfall design

5.8.1 Offshore outfall

Offshore outfall for deep outfall conduits

Figure 5-1 shows a typical submerged offshore outfall structure for a deep tunnel outfall conduit. The structure is a capped radial flow structure, constructed in precast concrete and placed on a prepared granular foundation.

The design of the outfall structure is essentially the same as the design of the submerged offshore intake structure except that the flow is reversed. The design comments in Table 5-2 Detailed design of submerged offshore intake structure therefore apply to offshore outfalls with the exception that:

- there is no biocide dosing system or AFD system;
- the outfall screen only needs to be a coarse screen (say, 40 mm diameter bars at a pitch of 250 mm) to prevent accidental entry of, for example, marine mammals when the system is not operating;
- screen area (discharge velocity), submergence and sill height need to be designed to mitigate the environmental impact of the CW discharge and satisfy the requirements of the CW discharge consent;
- submergence needs only to be checked to avoid the outfall cap being exposed in the trough of a wave.

The number and diameter of deep outfall tunnels will probably be decided by the construction contractor. A single tunnel would be adequate for the operation of the power station. Inspection and maintenance of the tunnel could be carried out using underwater engineering techniques during one of the scheduled plant outages (long tunnels being subject to similar constraints as intake culverts). The number and diameter of the vertical shafts may be decided by the construction contractor but may also be dictated by the design of the outfall with respect to heat dispersion.

The minimum depth required for an offshore outfall is site-specific. Typically, a minimum depth of five to eight metres below LAT could be required. The position of an offshore outfall should be designed to disperse the excess CW heat as efficiently as possible with the minimum environmental impact and recirculation. The design of the position therefore depends on site-specific factors such as:

- bathymetry;
- regime of tidal levels and tidal currents;
- position of the CW intake;
- prevailing meteorological conditions.

Offshore outfall for shallow outfall conduits

If the offshore outfall conduits are constructed by the immersed tube technique, the outfall structures will probably be constructed as an integral part of the immersed tube units. The comments on the design of an offshore outfall structure for deep outfall conduits apply equally to an offshore outfall structure for shallow outfall conduits except that isolating gates are not required. The need for isolating gates does not apply to shallowly-buried conduits because they cannot be dewatered owing to flotation.

A key aspect of the immersed tube design is that major temporary works are required in the inter-tidal zone where the offshore construction method changes to the onshore construction method (Figure 5-2).These works can have an environmental impact.

5.8.2 Inshore outfall

Inshore outfalls are usually situated in exposed locations to aid dispersion of the CW discharge. The outfall structure is either an open channel or a closed conduit that carries the CW flow across the foreshore and discharges it beyond the low water mark, Figure 4-4 and Figure 4-9 . Essentially the civil engineering structure will be subjected to coastal processes and will need to be designed to take account of tidal range, current regime and wave action (Figure 5-10). Suspended sediment is not a problem because the CW flow makes the structure self-cleansing. A key aspect of the inshore outfall is the permanent environmental impact on the foreshore.

Figure 5-10 Inshore outfall at Vasilikos, Cyprus

5.9 Heat dispersion

5.9.1 General process

The warm water from a CW outfall is of lower density than the cooler receiving water and generally forms a surface 'plume' in the receiving water. The excess heat in the plume is then gradually dispersed by cooling to the atmosphere. The formation of the plume and process of heat dispersion may be divided into three stages defined by the dominant physical process: near-field, mid-field, and far-field.

In the near-field, the behaviour of the discharge is governed by its initial momentum and buoyancy. The discharge is immediately cooled by entraining the less turbulent ambient water. The extent of cooling achieved by this initial mixing is governed by the design of the outfall. Design variables include:

- total number of discharge ports;
- discharge velocity through each discharge port;
- level of each discharge port relative to the seabed;
- direction of each discharge port relative to the tidal flow.

Another important factor is the temperature rise (ΔT) across the plant, higher values creating more plume buoyancy.

As the plume moves into the mid-field, it has usually reached the surface and the design of the outfall structure has less influence. In the mid-field, dilution of the warm water continues by turbulent mixing at the boundary of the plume. Buoyancy forces also continue to cause horizontal spreading of the plume. The plan shape of the plume is dominated by the ambient tidal current and any wind-driven current. In the mid-field, heat is dispersed partly by mixing and partly by cooling to the atmosphere.

In the far-field, the temperature difference between the plume and the receiving water is so small that buoyancy forces are negligible. The heat in the far-field is dispersed by cooling to the atmosphere and by residual currents. The far-field is sometimes known as the 'long-term heat field'.

'Primary' recirculation occurs if warm water from the mid-field plume is drawn into the CW intake. 'Secondary' recirculation occurs if water from the far-field reaches the CW intake. Secondary recirculation is not usually significant.

5.9.2 Methods of predicting heat dispersion

It is possible to predict heat dispersion with a physical model (Figure 4-10) but it is usually quicker and more cost-effective to use a numerical model. Numerical models for water quality and heat field calculations are presented in Section 6.3.4.

5.9.3 Dispersion for UK nuclear power stations

Each discharge plume for the next generation of nuclear power station sites will be designed to meet the requirements of the discharge consent. However, the shape and extent of the discharge plume will be site-specific because it will depend on unique site characteristics such as bathymetry, tidal range, current regime, ecology, meteorology and the detail design of the outfall structure.

An indication of the heat dispersion achievable may be obtained from model studies carried out for Sizewell B, CEGB (1986). Figure 5-11 shows the layout of the offshore intakes and outfalls for Sizewell A and Sizewell B. Figure 5-12 shows typical predicted surface temperatures for Sizewell B when the tidal current is flowing from north to south. The plume in Figure 5-12 is typical of heat dispersion in shallow water where the tidal currents are weak. The plume covers a very large area and is attached to the shoreline. In addition the mid-field plume passes directly over the intake for Sizewell A.

In order for the plume not to be attached to the shoreline, the outfall would need to be in deeper water (further offshore) and the tidal currents would need to be stronger. Even if a plume is not attached to the shoreline, it could still be attached to the seabed local to the outfall structure if the ΔT (and hence plume buoyancy) were insufficient.

To ensure efficient plume formation and heat dispersion, the outfall needs to be sited where the depth of water is sufficient to avoid attachment to the seabed. In other words, the natural ambient flow reaching the outfall should be sufficient to feed the entrainment flow demanded by the outfall without distorting the ambient regime.

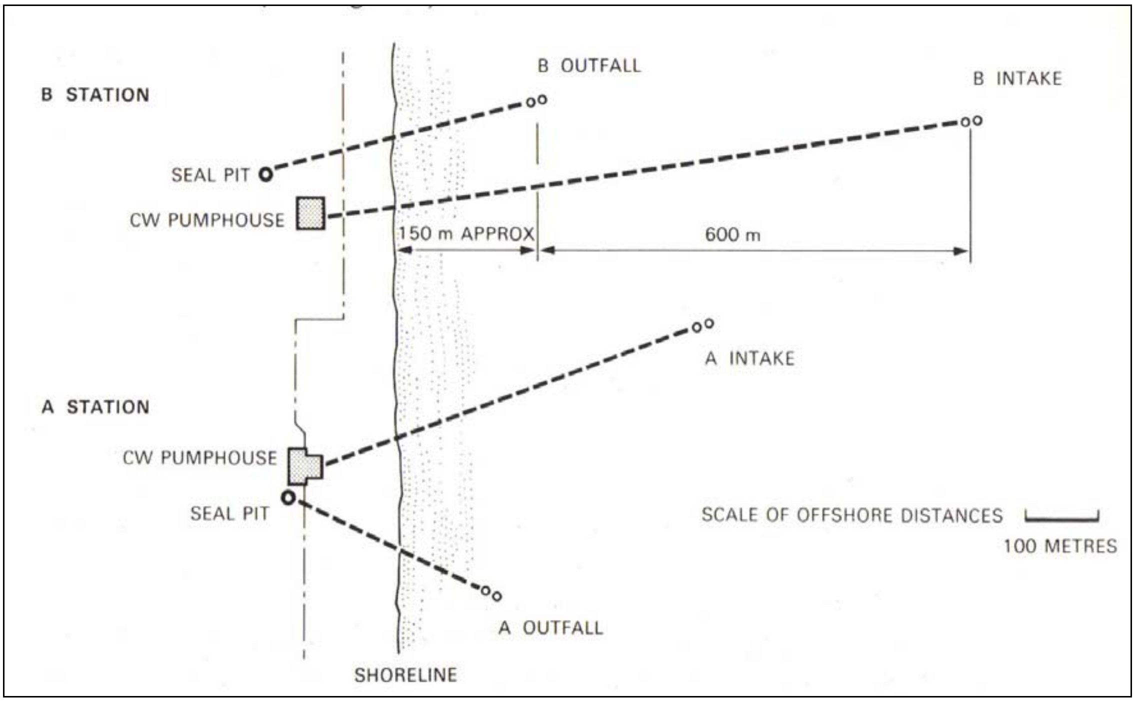


Figure 5-11 Layout of Sizewell A and Sizewell B

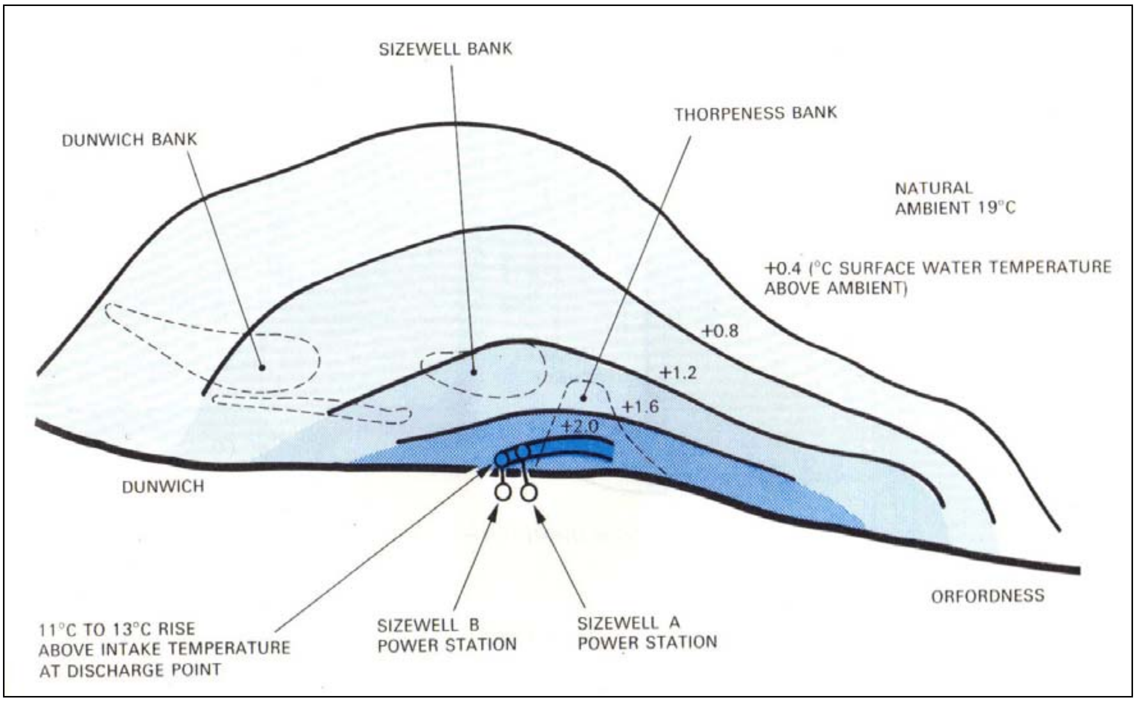


Figure 5-12 Temperature drop with distance from discharge point for Sizewell B

To avoid primary recirculation, the intake needs to be sited where the depth of water is sufficient to allow the ambient flow to feed the intake without distorting the ambient regime significantly and drawing in water from the discharge plume.

The heat dispersion and hence the environmental impact for a CW outfall can only be established by carrying out a comprehensive model study.

5.10 Liquid radioactive waste disposals with the CW discharge

Planned discharges

Radioactivity associated with the nuclear fuel cycle is almost entirely created within the reactor. Most of the activity is contained within the fuel cladding, but some is also created from neutron activation of the primary coolant and reactor components. Small quantities of fission products may escape from the fuel and these, together with some neutron activation products, appear in gaseous, liquid and solid wastes within the plant. Under the Radioactive Substances Act 1993, the Environment Agency needs to ensure that radiation exposure of members of the public from disposals of radioactive waste, including discharges, are as low as reasonably achievable (ALARA) by requiring new nuclear power stations to use the best available techniques (BAT) to meet high environmental standards (EA 2006a,b). This will help ensure that radioactive wastes and discharges from any new UK nuclear power stations are minimized and do not exceed those of comparable power stations across the world.

Radioactive waste management systems for the two designs currently undergoing generic design assessment may be seen on the requesting parties' websites, accessed through the joint regulators website[1]. Parties' submissions currently assume that all routine authorised liquid low-level radioactive wastes will be disposed of to the sea or estuaries together with the CW, thereby providing enormous initial dilution. However, the liquid radioactive wastes will first be held in tanks where they will be monitored to ensure compliance with disposal conditions before release to the CW outfall.

Accidental releases

The design of the cooling system has an important role in avoiding accidental releases. Design policy ensures containment by means of a series of physical barriers, so that there is little opportunity for any accidental release of radioactive waste on site to find its way into the CW system. Also, from the CW pumps to the condensers and heat exchangers the cooling water is under positive pressure, preventing ingress in these sections. In a directly cooled system, the only free surfaces where contamination could enter are at the CW pump forebays and the seal pit. With wet towers, the cooling water is under pressure from the CW pumps right up to the sprays inside the towers.

It is of interest to consider whether alternative cooling system designs would lead to different outcomes in the event of radionuclide leakage into the CW circuit. With cooling towers, huge surface areas would need to be decontaminated and there is also a risk of spreading radioactive material over the countryside if the incident were to coincide with high winds and/or damaged or displaced eliminators. On the other hand the contamination would be captured on site within a relatively small volume of water, assuming that the purge were stopped and the reactor shut down.

With direct cooling the contamination would be dispersed in a large volume of water. The type of outfall, whether surface- or bottom-opening, and the local hydrographic conditions are likely to influence the degree of dispersion of any radionuclide particles in the cooling water.

There should be no off-site risk with air-cooled (dry tower) stations since there is no CW discharge and no contact between cooling water and air. With an air-cooled condenser, contamination in the steam circuit (as from a BWR) would be transferred to the large surfaces of the condensers. However, any leaks in the pipework between the turbine and condenser and in the condenser itself would be inward. There is usually a small amount of water-cooling for the auxiliary circuits at air-cooled stations, but this is small volume and retainable.

6 Environmental issues associated with cooling

6.1 Effects associated with abstraction

6.1.1 Entrapment, entrainment and impingement

The abstraction of surface water from natural sources means that organisms present in the source water will be drawn into the water intakes. These organisms can include anything from planktonic bacteria and algae to macroinvertebrates and fish; much more rarely, aquatic mammals and diving birds can enter and become trapped. Smaller organisms enter involuntarily, being at the mercy of prevailing water currents. Some larger vertebrate animals enter either because they are already sick or moribund or because they have become disorientated, for example in the dark or in very turbid waters. In some cases, larger predators such as bass (*Dicentrarchus labrax*) enter opportunistically to feed on whitebait and other organisms that become concentrated within the screenwells of power stations. An experiment at Fawley Power Station (Southampton Water, Hants.) in which large marked rainbow trout were released into the screenwells revealed that they were recaptured in screening samples over the next six months. On inspection, they were found to have gained in weight considerably and had stomachs bulging with small fish (Turnpenny and Holmes, unpublished).

Of organisms entering with a power station's cooling water, some may enter and leave at will, while others will become impinged on the screen meshes of the filtration system. The remainder will penetrate the screen meshes and enter the main cooling system. The following definitions are commonly used:

- Entrapment - inadvertent entry into the CW system of aquatic organisms caused by the ingress of water; the term implies that the organism is unable to resist capture, owing to poor or no swimming ability, or to failure to detect the water intake.
- Impingement - retention of entrapped organisms on CW intake screens employed to prevent debris entering the CW heat exchangers; to become impinged, organisms must be large enough to be retained by the screen meshes (usually includes e.g. juvenile-adult fish, macroinvertebrates such as shrimps, crabs and large molluscs and marine algae).
- Entrainment - passage of entrapped organisms that penetrate CW screens (typically zooplankton including ichthyoplankton and phytoplankton), via the pumps, heat exchangers and other components of the CW circuit and back to the receiving water. Note: the size break-point between impingement and entrainment depends on the size of mesh openings in the CW screens.

6.1.2 Legal requirement for fish screening

An assessment of legal requirements is described by Turnpenny and O'Keeffe (2005). These arise out of the Salmon and Freshwater Fisheries Act (SFFA) 1975 as amended by the Environment Act 1995. Measures within SFFA Sections 14 and 15 apply solely to the migratory salmonids, Atlantic salmon (*Salmo salar*) and sea trout (*Salmo trutta*) and technically apply to waters frequented by these species, a term which is interpreted to require proof that there is a self-supporting population of at least one of these species present, rather than one maintained by stocking.

In order to comply with the Water Framework Directive (WFD), Fish Passage and Screening regulations were proposed. These required the consideration of whether additional measures would be needed to ensure all species of fish are protected rather than just salmonids alone to achieve good ecological status. However, following recent consultation [22], the Better Regulation Executive has put development of these regulations on hold until 2011 due to the potential economic impact on business. It should be noted that, despite these delays, any current and future power station developments would need to adhere to the Fish Passage and Screening regulations in order to comply with the WFD.

In addition the Eels (England and Wales) regulations (2009) came into force in Jan 2010. The regulations reflect European wide concern over the collapse of the European eel *Anguilla Anguilla* population and require the provision of screening to be considered for eels.

In Natura 2000 sites, operators and developers are required to develop mitigation measures and monitoring programmes to ensure that conservation objectives of the site are not jeopardized.

All of the above fish screening regulations are relevant regardless of whether or not an abstraction licence is required at a site.

6.1.3 Impingement of fish and other biota

Screen mesh sizes

Screen mesh size is the main factor determining the sizes of organisms retained on the CW drum or band screens and those that become entrained. For fish, the size that will be retained by the screen is a function of mesh size and fineness ratio (body length divided by maximum body diameter), the latter being high for elongate, thin fish such as eel or pipefish and low for rotund ones such as lumpfish (Figure 6-1). Older UK power stations of CEGB vintage, including all existing nuclear plants, typically use a mesh of 8-10 mm square aperture. These would retain for, example, bass larger than about 70 mm standard length[23] and eels longer than 240 mm. More modern UK stations fitted with purpose-built fish return systems are required to use mesh sizes of 6 mm or less under present guidance (Turnpenny and O'Keeffe, 2005), which would reduce the bass and eel thresholds to around 30 and 100 mm respectively. That is not to say that in either case all fish below these sizes would pass through and become entrained, as some will strike the screen sideways-on and become impinged.

[22] http://www.defra.gov.uk/corporate/consult/fisheries-legislation/ (viewed 22/02/09)

[23] Standard length is the length from the tip of the snout to the caudal peduncle. It is commonly used in power plant studies, rather than total or fork length, as tail fins are often damaged as a result of screen handling on older non-fish-friendly screens.

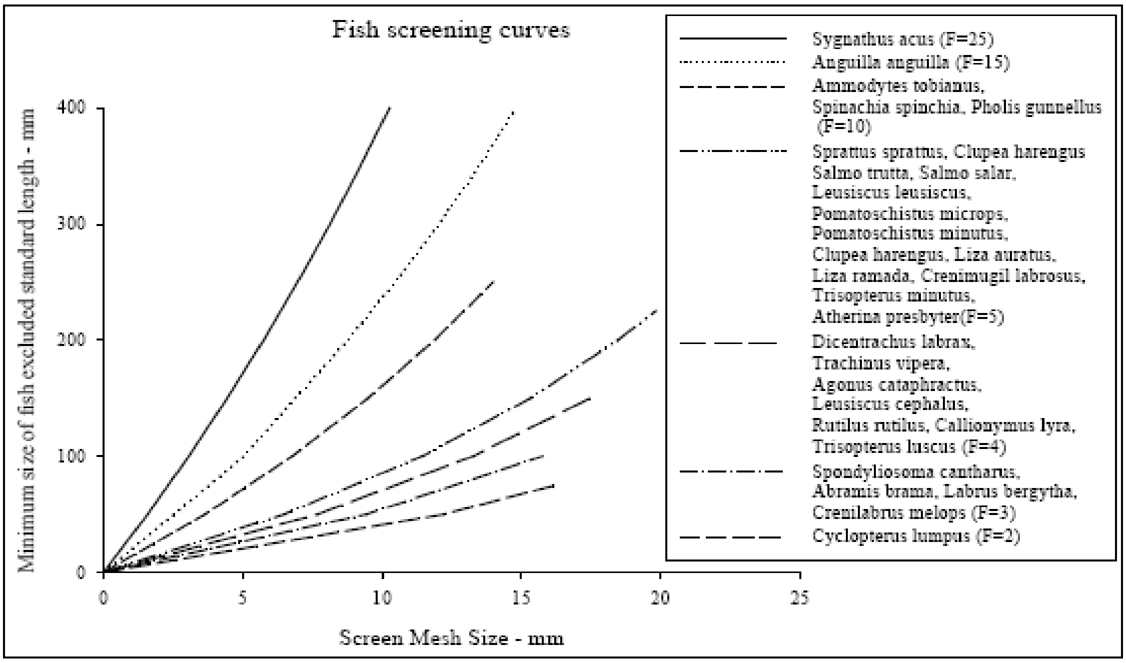


Figure 6-1 Mesh size curves for screening fish of different body shapes (from Turnpenny and O'Keeffe, 2005, after Turnpenny, 1981)

Finer mesh sizes are more common in continental Europe. French nuclear stations operated by EDF favour a three to four mm plastic mesh, which reduces the entrained component to smaller plankton, including fish eggs, larvae and postlarvae. The use of finer meshes may or may not be more protective of fish, depending on the comparative survivability of the impingement and entrainment processes, and in the case of impingement, whether the impinged material is returned to the source water body. These aspects will be discussed further in this and following sections.

Fish inundation and the process of fish removal by screens

The process of impingement, once within the screenwell, is rapid for non-motile or weakly swimming organisms but may be protracted for many fish. The natural reaction of fish held in a current is to attempt to swim against the flow. Investigations of fish impingement patterns in tidal systems suggest that even small fish such as sprats (*Sprattus sprattus*) are capable of active avoidance of impingement for several hours, usually until the water level in the screenwell drops and higher velocities force them onto the screen meshes (Turnpenny and Utting, 1981). This behaviour increases the peak loading of fish onto the screens leading up to the low water period and has been a key factor causing screens to block with winter sprat inundations, notably at Sizewell A and the Dungeness stations. One CEGB proposal involved the installation of mesh baffles within the screenwell to reduce the volume of water accessible to fish and thus force them quickly onto the screens, thereby smoothing out the screen loading over the tidal cycle. This was never implemented.

Sudden sprat inundations are caused when large overwintering schools move close inshore. These remain an operational risk at eastern and south-eastern UK open coastal sites. Historically, clupeid inundations, including herring (*Clupea harengus*), have also affected sites on Scottish east-coast estuaries such as Longannet and Kincardine Bridge. Other pelagic species may have a similar potential to cause problems and recent observations of large concentrations of anchovy (*Engraulis encrasicolus*) eggs and larvae off the Blackwater Estuary on the south-east coast of

England are noteworthy (Andy Payne, Cefas, personal communication 2008 surveys). This species is currently at the northern end of its distribution at this latitude.

While drum screens are rotated continuously, and therefore lift out debris relatively soon after its impingement on the screen, this is not always true of band screens. Band screens have many articulating parts and therefore inherently wear out more quickly, especially in abrasive, sediment-loaded waters. To extend band screen life, they are often rotated only for brief periods after the accumulation of debris on the screen face has caused partial blockage and a pressure differential is detected by instrumentation. Consequently, fish may be pinned against the screens for several hours before removal.

Band and drum screens usually have more than one speed setting. Normal operation uses the lower speed settings to conserve the life of the bearings, tripping to higher speeds only during periods of high debris loading.

Material is removed from drum or band screens, as the ascending face of the screen emerges through the water surface, by a ledge or 'trash elevator' onto which they slide or fall (Figure 6-2) Unless specifically designed for fish recovery, these elevators are normally open-ended and free draining, so that biota come into hard contact with the screen structure. This exacerbates injury risk. While this may be immaterial when organisms are destined only for landfill disposal, it becomes an important consideration where screens not originally designed for safe fish handling are subsequently adapted for fish return.

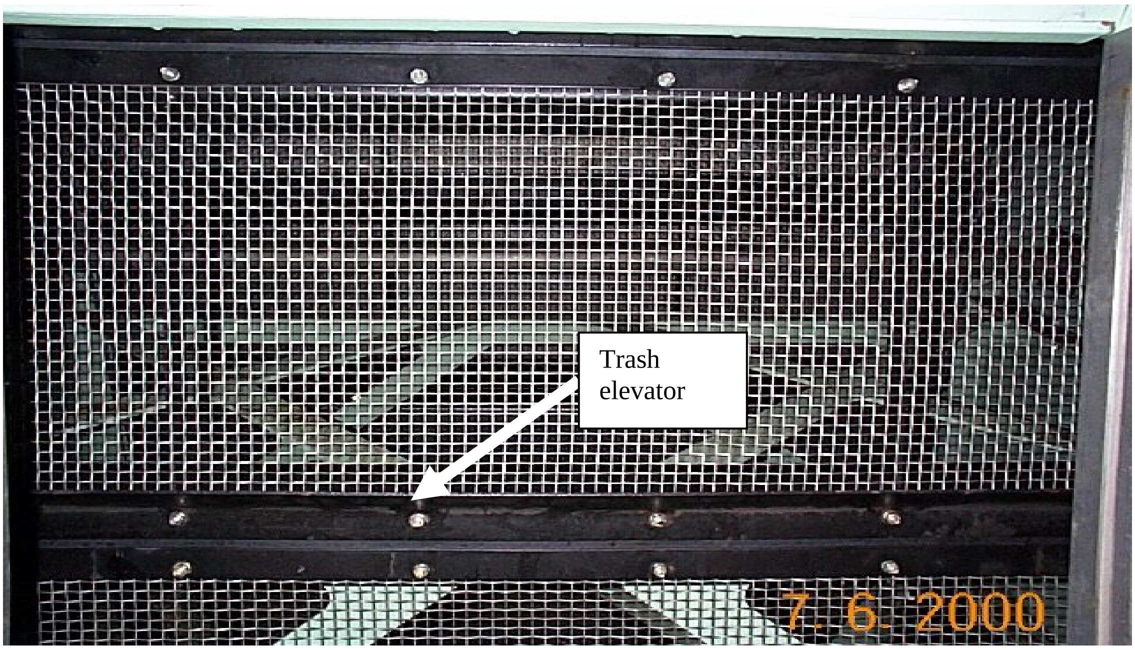


Figure 6-2 Bandscreen panel fitted with six-mm square woven stainless-steel mesh. The trash elevator is in the form of a narrow ledge which does not retain water and is unlikely to properly retain larger or writhing fish such as eels.

Quantities and types of organisms impinged at UK stations

The value of power station screens as fish sampling tools was recognised in the 1960s when biologists first saw fish returning to the tidal Thames through monitoring the contents of trash baskets (Wheeler, 1979). This prompted university biologists to do the same elsewhere, for example on the Severnside and South Wales stations (Claridge and Potter, 1985; Caridge, Potter and Hardisty, 1986). From the 1970s, the National Rivers Authority began regular sampling of fish from West Thurrock power station to monitor for disease and contamination (see Power, Attrill and Thomas, 2000) and continued until the station's closure in 1992. MAFF Directorate of Fisheries Research (DFR – now Cefas, Lowestoft) also recognised the value of this method to assess year-class-strength in bass (*Dicentrarchus labrax*), principally at Oldbury-on-Severn and Kingsnorth power stations (Pickett and Pawson, 1994), to cover west and east coast sub-stocks respectively. Meanwhile, by the mid-1970s, the CEGB was concerned by legal actions being launched against power companies in the USA. A successful prosecution of Indian Point power plant on the Hudson River (New York) in 1972 led to a fine of US$1.6 million resulting from the capture of large numbers of small fish in power plant cooling water screens (Langford, 1983). The CEGB established its own programme of fish impingement studies via the staff of its Fawley Marine Laboratory. As the CEGB was a publicly owned body, many of these were carried out in collaboration with MAFF DFR (see Langford and Utting, 1977; Turnpenny *et al*, 1988).

Joint CEGB/MAFF investigations carried out at Sizewell A Power Station as part of the Sizewell B pre-application work still provide the benchmark for fish impingement survey design and analysis (Turnpenny and Taylor, 2000). Fish impingement became one of the key non-nuclear environmental issues for Sizewell B, after local fishermen objected to catches of mainly undersized fish by the A-station. A comprehensive assessment was therefore undertaken, the first fully quantitative one of its kind. The study collected complete catch over 24-hour periods from all operating screens on 40 dates per year, with ten randomly selected dates per quarter. This 'systematic randomised' sampling was used to obtain representative samples. Taking 24-hour samples eliminates diurnal and short-term tidal cycle influences, while randomisation takes out longer-term tidal cycle effects and any systematic tendency in terms of CW abstraction pattern. The number of 24-hour samples required may vary between sites according to variability and required statistical power, but experience shows this to be a good default. More details of the methods and findings from the studies carried out at Sizewell A and B can be found in Turnpenny and Taylor (2000).

The sampling method for such studies is straightforward. Usually, the trash baskets are cleared and net liners are placed inside at the start of the sampling period. Fish and other biota are then removed at intervals and sorted to species, identified, weighed and measured as required (Figure 6-3). Samples may be retained for laboratory analysis of pesticide residues, stomach contents and so on.

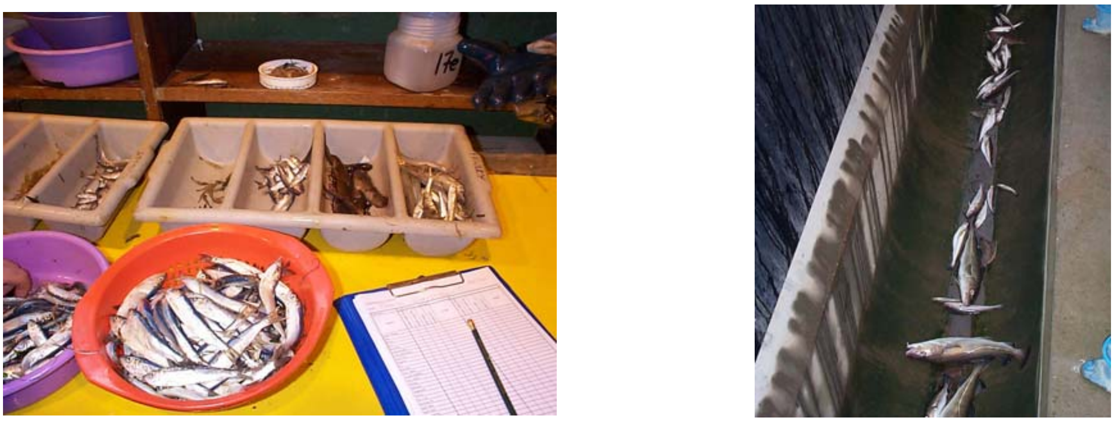


Figure 6-3 Sorted sample of fish collected from power station screens (left). Catches at saltwater sites typically contain a wide range of species but are often dominated by pelagics, such as sprat and herring. The bulk of the catch is often made up of juveniles or smaller species of under 20 cm in length, although certain locations and intake designs can put larger fish at risk (right).

Indicative catch rates by impingement for British and some Channel-coast French sites were given by Turnpenny and Coughlan (2003) and ranged from a few tonnes per year to the highest recorded value of 240 tonnes per year at Gravelines in France. More useful are figures for the catch per million cubic metres of CW flow, which take out the effect of plant size. These still show order-of-magnitude differences between sites, which are due to factors such as intake design and locality. The highest impingement rates tend to occur at sites on the open coast, such as Sizewell and Dungeness, owing principally to the abundance of pelagic fish which dominate the catches in these areas. However, there are always exceptions, such as Wylfa, sited on the rocky north-west point of Anglesey, where catches are exceptionally low.

Figure 6-4 Coarse screen blocked by fouling

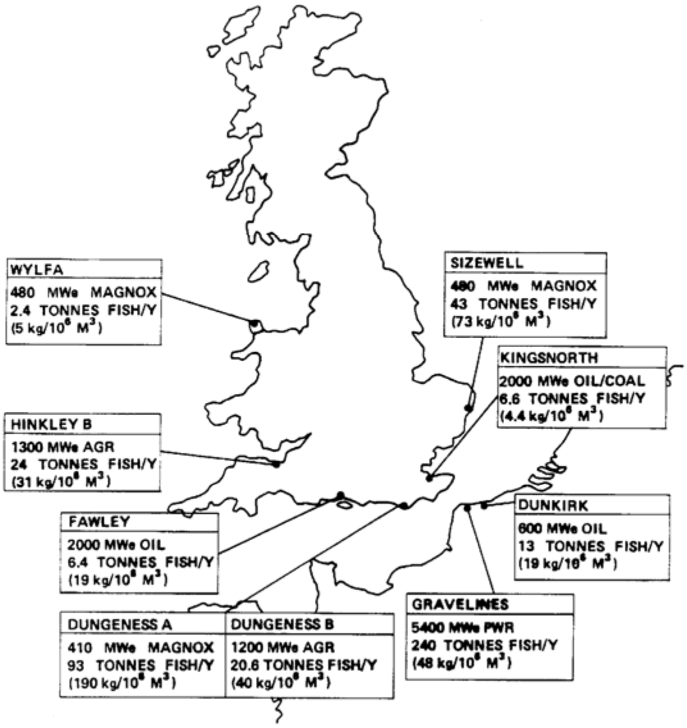


Figure 6-5 Estimated annual total quantities of fish impinged at UK estuarine and coastal power stations (Turnpenny and Coughlan, 1992)

Government plans in the 1980s to expand nuclear generation following the successful licensing of the Sizewell B PWR led to an urgent need to screen coastal sites for new nuclear build. The scientific approach to fish impingement at Sizewell had proved successful in allaying the fears of commercial fishermen and it was decided the same approach should be adopted for other proposed nuclear sites. This was straightforward at existing nuclear sites where screens could be sampled directly, but would prove more difficult at greenfield sites. To bridge this gap, CEGB Fawley Marine Laboratory developed an expert system model capable of predicting order-of-magnitude impingement rates by analyzing patterns observed at operating sites. PISCES (Prediction of Inshore Saline Communities Expert System) enables the numerical abundance, seasonality, age composition and biomass of the most common impinged fish and crustaceans to be predicted for any British coastal or estuarine site, given basic information such as location, salinity, substrate types and adjacent habitat types. It is based on some twenty years of research into impingement rates at coastal power stations, which has led to the development of ecological rule-sets that enable predictions for unsurveyed sites. PISCES has been widely used in relation to new plant applications in British and northern European (Dutch and French) power stations, either in the absence of or to supplement data. PISCES has shortcomings, being based on old CW intake designs which would not conform to modern design requirements, but a new version is currently being developed by Pisces Conservation Ltd (Seaby and Henderson, 2009).

A variety of crustaceans is usually found in screen samples, including crabs, shrimps and prawns of various kinds. At Sizewell A, for example, Whitehouse (1986) estimated an annual impingement catch of the brown shrimp, *Crangon crangon,* of 16 tonnes. At Hinkley Point, catches regularly include quantities of *C. crangon*, along with the pelagic prawn, *Pasiphaea sivado*, and the edible prawn, *Palaemon serratus* (Henderson and Holmes, 1982). Generally, crustaceans are resistant to handling and where FRR systems are provided (Section 6.1.6), the majority can be returned unharmed.

The largest invertebrate species impinged on power station intake screens are cephalopod molluscs (squid and cuttlefish). The impingement and population dynamics of cephalopods was studied at Fawley Power Station, Hampshire, between 1978 and 1981 by Bamber (1981). Five species were recorded, three of which, *Loligo vulgaris*, *L. forbesi* and *Sepiola atlantica* were either too infrequent or not adequately represented for analysis owing to sampling selectivity. The cuttlefish *Sepia officinalis* (a species of commercial value) and the small squid *Alloteuthis subulata* were common and their local life histories were analyzed. Impingement was seasonal, occurring between April and the onset of winter, when temperatures were above 10ºC: adults arrived during the early phase, to breed and then die (both species being annual), to be replaced by their young as they grew on, until finally migrating offshore as water temperatures dropped. The quantities involved in impingement – 0.24 tonne per year – were not perceived to be a problem to power station operation, or to be significant to local cephalopod stocks. Most of the impinged biomass was from post-breeding, senescent adults.

Injuries and survivability of impingement on traditional drum and band screens

The proportion of fish, crustacea and other organisms surviving impingement was practically zero at almost all UK estuarine and coastal power stations prior to the 1970s, as it was common practice to collect all the screen backwashings in trash baskets and dispose of them to landfill; this remains the practice at some. At a few stations (such as Bradwell, Sizewell A), the material was macerated and discharged back to sea, at least in this way putting some material back into the marine food web.

Indirectly-cooled inland stations more commonly put backwashed material back to the river. In cases where material is returned to the source water, the practice may require a discharge consent from the Environment Agency. Improved fish protection technology at newer stations has greatly reduced the need for waste disposal from backwashings (see Section 6.1.3).

A number of studies have investigated survival rates of fish collected from CW screen backwashes. Such studies are usually done to assess the benefits of returning the fish to the wild. Depending on the CW intake arrangement, fish collected at this point will normally have experienced a number of possible stress factors, as a result of:

- prolonged swimming ahead of the intake coarse screens to avoid entrapment;
- passing through the intake opening and any canal or culvert leading into the plant, particularly in deep tunnels where fish may experience rapid pressure fluxes in the ascending and descending limbs;
- turbulence in the forebay and screenwells;
- biocide toxicity (normally applied upstream of the screening plant);
- prolonged swimming to avoid becoming impinged on the drum or bandscreens;
- being pinned on the screens prior to removal (may be for hours in the case of band screens that only rotate in response to increased hydraulic head-loss);
- repeated recycling through the screenwell when screen elevators (also known as ‘troughs’ or ‘buckets’) are inefficient at retaining fish: especially a problem for large fish or sinuous fish such as eels;
- exposure to high-pressure backwash sprays;
- prolonged retention in screen hoppers and launders caused by inadequate backwash spray flow or blockage of paths by accumulating weed and debris.

The test protocol usually involves collecting fish in nets or baskets placed beneath outlets from the screen launders, separating, identifying and counting live versus dead fish at the time of removal of fish from the launders. This is followed by monitoring survival rates of live fish over the subsequent hours or days. All fish are also examined for external injuries, such as scale loss, fin damage, flesh wounds and eye injuries, then for internal injuries such as swimbladder rupture or haemorrhaging.

A number of injury and survivability studies are known to have been carried out at UK power stations, although data have generally not been published and are not easy to track down. A brief study carried out at Fawley Power Station (Hampshire), which has drum screens that have no modifications to make them fish-friendly, revealed that over 90 per cent of fish exhibited external signs of injury. Commonest symptoms were fin damage (splitting or haemorrhaging), scale loss and eye damage (Turnpenny, 1992). Larger fish, which can fall off the screens many times before being successfully removed, will often suffer abrasion sores, gashes or, in the case of eels, spinal fracture (Figure 6-6).

Figure 6-6 Examples of fish injuries caused by impingement. Left: bleeding into the eye is common as a result of exposure to pressure change. Right: an eel with spinal fracture.

Some studies have included dissection to identify swimbladder rupture, a symptom caused by rapid depressurisation, which can cause the swimbladder gases to expand and tear the surrounding tissue. A similar effect is seen in fish that have passed through hydroelectric turbines (Turnpenny, 1998). Gadoid fish such as whiting (*Merlangius merlangus*) and pout (*Trisopterus luscus*), which regulate swimbladder volume by vascular exchange (known as “physoclists”) as opposed to by gulping and venting air via the gut (“physostomes”), are particularly susceptible. At Sizewell B, more than a third of whiting and pout collected from the screens were found to have ruptured swimbladders. While swimbladder injuries will heal within a few days (Turnpenny *et al.* 1992), loss of swimbladder function in the meantime may affect functions like buoyancy, balance and hearing, potentially putting fish at greater risk of predation. Station designs most likely to cause pressure effects of this kind are those in which the CW tunnels descend deep below the water surface, so that fish are exposed to rapid depressurisation as they are brought back up to the surface and out onto the screens. Under these conditions, outgassing of the body fluids can also occur, causing symptoms similar to “the bends” in humans.

Fish survival rates post-impingement have generally been estimated only following some design modification to improve survivorship. Turnpenny (1992) reviewed findings from a number of fish return systems in Europe and the USA, which revealed marked distinctions between categories of fish. Pelagic fish, often with ‘deciduous’ scales, in normal life will avoid contact with the bed and hard structures and are unlikely to survive impingement. Demersal fish are more robust and can usually cope with some physical contact but can be susceptible to swimbladder rupture; these have a moderate chance of survival. Benthic species, those normally living in contact with the sea or river bed, are physically robust and lack a significant swimbladder, and generally have the best survival rates (Table 6-1). Monitoring of post-impingement fish and crustacean survival rates was carried out routinely between 2000 and 2003 by Clough *et al.* (2003) as part of post-commissioning studies at Shoreham power station (W. Sussex). Overnight (16-24 hour) survival rates averaged 80 per cent for *C. crangon* and 76 per cent for prawns (*Palaemon serratus*). The system at Shoreham comprises a standard Brackett-Green bandscreen with traditional fish buckets.

Best practice in design of drum and band screen systems for fish return is discussed further in Section 6.1.5.

Table 6-1 Typical fish survival reported from studies of drum or band screens with simple modifications for fish return (Turnpenny and O'Keeffe, 2005).

Fish group	Survival rate 48 hours after impingement
PELAGIC e.g. herring, sprat, smelt	Less than 10%
DEMERSAL e.g. cod, whiting, gurnards	50-80%
EPIBENTHIC e.g. flatfish, gobies, rocklings, dragonets, and crustacean	More than 80%

6.1.4 Entrainment of fish and other biota

Measurement of entrainment rates

Compared to impingement, fewer studies of fish entrainment rates at UK power stations have been reported. Entrainment is less visually obvious than impingement and sampling techniques are more difficult. For fish species, losses of eggs and fry are also thought to be less important to maintenance of the stock than the capture of more mature fish, which represent a higher value to the population. However, development in recent years of effective mitigation techniques against impingement rates and mortalities now makes losses from entrainment a bigger part of the environmental impact of CW abstraction.

Entrainment rates are measured by intercepting a sample of CW flow with fine-meshed plankton nets, typically of 0.275-0.5 mm mesh-size. This is achieved by:

- a plankton net placed in part of the intake flow (for example in the forebay, where the degree of turbulence allows);
- drawing water from a tapping on the pressure side of the main CW pumps or screen-washwater pumps (which usually draw from the screenwells) and passing it through a suspended plankton net;
- using a purpose-built powered plankton-sampler lowered into the forebay (Coughlan and Fleming, 1978).

When a power station is yet to be built and there are no historical data from a previous or existing plant, prediction of entrainment impacts usually relies on conducting plankton surveys in the vicinity of the proposed intake point, perhaps also backed up by sampling a reference site that is expected to remain undisturbed after the development. Standard plankton-sampling methods commonly involve oblique or undulating hauls to ensure that most of the water column is sampled. This overcomes the risk of missing populations that may be found only at particular depths, for example where the water column is stratified owing to temperature or salinity effects. Some types of power station intake structure abstract selectively from deeper layers to reduce recirculation of the buoyant plume and therefore do not draw in a sample representative of the whole water column. This may explain observations reported by Coughlan and Davis (1980) for Bradwell Power Station (Blackwater Estuary, Essex) and Dempsey (1988) for Fawley Power Station (Southampton Water, Hampshire), which in both cases show concentrations of entrained ichthyoplankton to be an order of magnitude less than those found in the open water. In any event, despite the tidal rise and fall at coastal or estuarine sites exposing the CW intake orifice to most of the vertical range of the water column, it is rare that the CW water will be abstracted from the full vertical range.

A key factor in the success of these sampling methods is achieving adequate sample volumes and good diel and seasonal coverage. It is now common for the Environment Agency to attach monitoring conditions to new abstraction licences and this has led to the need for sampling protocols for, *inter alia*, measurement of entrainment rates. While no formal protocol has been established, developing practice at new sites has been to specify 24-hour sampling periods using sample flow rates of 10 to 25 ls^{-1}, where these flow rates are practically achievable. This appears to provide good representation of fish taxa, which are notoriously patchy within the plankton and may otherwise be missed. Sampling is typically carried out at least monthly (sometimes weekly during periods of peak ichthyoplankton activity) all year round, or for a six-month period from spring to late summer, outside which fish leave the planktonic phase.

Other types of plankton entrained will include temporary (meroplankton) and permanent (holoplankton) planktonic taxa, including molluscs, crustaceans and other invertebrates, as well as phytoplankton. Holoplankton are commonly more abundant than ichthyoplankton, and adequate samples can be obtained by the methods outlined above or from much smaller “bucket” samples. However, these species show marked (and well-documented) seasonality in their abundance in the plankton. Meroplankton include larvae of a wide range of taxa which have a sessile, benthic or pelagic adult life; many of the invertebrate meroplankters are of commercial significance (larvae of commercial species of shellfish and crustaceans). These species also show constrained seasonality in their presence in the plankton. Again, it must be appreciated that these taxa are patchy and are stratified in the water column.

Entrainment at UK inland stations

Studies at inland stations have been limited and confined to investigation of larval fish entrainment. A brief study at the indirectly-cooled Didcot A power station was carried out by Aston and Fleming (1992) and was the first of its type in the UK. The station was estimated to entrain around 1.9×10^6 fish fry annually for a CW abstraction rate of 2.4 m^3s^{-1}; however, this value did not fully take into account the seasonal variability of entrainment and was therefore considered to be an overestimate.

A subsequent investigation at Ratcliffe-on-Soar Power Station (CW flow 2.08 m^3s^{-1}) on the River Trent undertaken by Smith (1998) between 1994 and 1997 revealed a strongly seasonal pattern of fish entrainment. This was characterized by influxes of newly hatched ‘pinhead’ fry of coarse fish during the spring and early summer months following spawning, quantities declining throughout the summer as fry numbers in the river decreased owing to high natural mortality rates. Greater ability of fish to avoid entrapment is likely to have been a factor as the season progressed and fish grew larger. The overall loss rate of fish averaged $3.45\text{-}7.98 \times 10^5$ fry per annum over the three years. Dominant species were roach (*Rutilus rutilus*), bream (*Abramis brama*), bleak (*Alburnus alburnus*) and chub (*Leuciscus cephalus*). Smith converted these numbers to equivalent adult values (EAVs) which amounted to 2,290 adults per annum (see later in this section for a fuller description of the EAV methodology).

Studies of entrainment patterns at other types of inland water-intake consolidate the picture from power station studies. Turnpenny (1999) carried out a desk study of the combined coarse-fish fry entrainment potential of all raw water intakes on freshwater

Thames (nine in total) based on an extrapolation of data from only one of the intakes. The study concluded that substantial numbers of young coarse fish were likely to be lost to the fishery. As a worst case, if all the intakes operated at maximum licensed capacity (around 80 m^3s^{-1} combined capacity), the loss of fish to entrainment could amount to the equivalent of up to 45 per cent of the adult standing stock of the Lower Thames. A subsequent two-year field study carried out in 2006-7 (Turnpenny *et al*, 2008) showed that even this figure may have been low and potential losses at maximum licensed capacity could amount to 61 per cent of the total adult stock.

All of these studies showed common features, which can be summarized as follows:

- Proportional losses of stock to entrainment are related to the abstraction flow; cumulatively along a river reach they can represent a substantial impact on numbers of fry available to recruit into the adult fishery.
- Catch rates are highly seasonal, peaking in the spring, shortly after spawning, but extending through the summer months.
- Entrainment rates tend to be positively correlated with river flows and are highest at night, corresponding with fry migration studies on large river systems which show that fry drift or migrate downstream principally under these conditions as part of their natural distribution mechanism (Pavlov *et al.* 1978).

Entrainment at UK estuarine and coastal stations

Survivability of entrainment in once-through cooling systems

In the absence of reliable evidence to the contrary, it has often been assumed that all plankton passing through a power station's CW circuit will be killed. This view is central to US EPA thinking and policy on entrainment impacts and is a key reason why US assessments of the impact of direct cooling systems on ecosystems yield substantial estimates of harm. However, there is convincing evidence from UK studies that survivorship in various taxa, including fish, other zooplankton and phytoplankton can be high. Moreover, as survivorship has been shown to be influenced by aspects of the design and operation of once-through cooling water systems, it is clear that one conclusion will not hold for all stations and that there may be opportunity for increasing survivorship through plant design and operational controls.

The belief that cooling circuits act like vast sterilizers stems from the fact that plankters are subject to a number of potentially lethal stressors during passage. These include:

- The same mechanical and hydraulic stresses in the intake line and forebay and screenwells as impinged organisms.
- More protracted exposure to potentially toxic biocide levels throughout the CW circuit.
- Rapid temperature increase through and beyond the condenser boxes.
- Changes in hydrostatic and hydrodynamic pressure caused by differences in level and by pumping.

- Hydraulic shear stress, turbulence and abrasion associated with passage through screens, culverts, small-bore condenser tubes and other pipework.

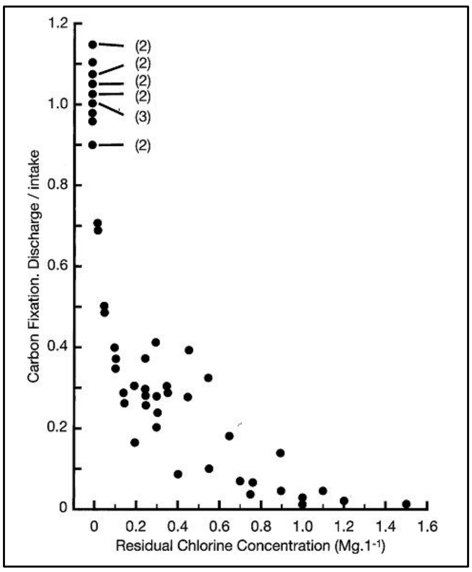
Three main approaches have been used to study the viability of organisms following exposure to entrainment stressors:

- power-plant-based entrainment monitoring, where live:dead ratios are compared between the CW plant inlet and outlet points (see Coughlan & Fleming, 1978; Coughlan & Davies, 1983; Dempsey, 1983);
- laboratory-based cooling-circuit simulation studies (see Bamber *et al.* 1994; Bamber & Seaby, 2004; Kennedy *et al.* 1974; Beck *et al.* 1975);
- laboratory dose-response studies using chlorine and temperature combinations.

Power plant-based entrainment monitoring

In the UK in the 1970s, studies were undertaken to examine the survival of entrained phytoplankton and zooplankton at five coastal/estuarine power stations: Fawley (Hampshire), Sizewell A (Suffolk), Kingsnorth (Kent), Bradwell (Essex) and Heysham 1 (Lancashire) (Coughlan & Davies, 1983). Subsequently, phytoplankton productivity at Fawley Power Station was measured to estimate the survival of entrained organisms (Davies, 1983). Seawater (10 litres per replicate) was sampled from the intake and outfall and incubated with the radioactive carbon isotope C^{14} for three hours under standard temperature and light conditions. The rate of C^{14} fixation was compared between intake and outfall samples to give an estimate of the survival rate of phytoplankton passing through the CW system. Phytoplankton productivity fell by 50-60 per cent having passed through the CW system under routine conditions at Fawley Power Station (ΔT 8-10°C, <0.2 mgl^{-1}Cl at outfall). The main cause of mortality was chlorine concentration. Experimentally varying the dosed chlorine level allowed the effect of chlorine toxicity to be assessed. The main cause of mortality (assuming that the rate of C^{14} fixation was a valid proxy of survival) was concluded to be exposure to the biocide.

Figure 6-7 Effect of chlorine concentration on carbon fixation by phytoplankton at Fawley Power Station, Hampshire (Source: Davis, 1983)



Inhibition of photosynthesis by phytoplankton has also been observed in chlorinated cooling water by researchers in estuarine water (Hamilton *et al.* 1970) and seawater (Carpenter *et al.* 1972; Khalanski, 1977).

Davis (1983) sampled seawater from the intake and outfall at Fawley Power Station (Hampshire, UK) and measured the temperature, salinity and primary productivity in all samples. The concentration of total residual chlorine was determined in all outfall samples. Primary productivity was reduced in the presence of chlorine, with a 96 per cent reduction at a concentration of one mg l^{-1}. Where final water temperature reached up to 23 °C, productivity was observed to increase by up to 15 per cent. Where final water temperatures exceeded 23 °C the productivity decreased by up to 11 per cent. No correlation was found between ΔT and productivity (Davis, 1983).

Two trials were carried out in the absence of chlorination and temperature rise (Davis, 1983), allowing an assessment of mechanical damage effects to be made; in both cases, primary productivity increased to the same order as that observed for the temperature rise. The change in productivity observed here was linked to the heterogeneity in sampling and observations reflecting the views of other researchers (Flemer & Sherk, 1977) that the mechanical effects of entrainment on phytoplankton are too small to be detected in field studies (Davies, 1983).

Davis (1983) acknowledged the wide variation in reported effects of power plant CW chlorination on the primary productivity of entrained phytoplankton; he attributed the variation to differences in location of the power plants, and variations in phytoplankton populations, water quality, operating conditions and sampling and/or experimental techniques where the studies took place.

Davis (1983) recognised other limitations in his study, such as holding phytoplankton samples in the chlorinated water for up to three hours post-sampling. In reality, on return to the sea, chlorine concentration becomes progressively lower as the discharge water becomes diluted. This was not achieved under laboratory holding conditions. Despite this, Davis was able to conclude that phytoplankton productivity was not as inhibited by chlorination as previous studies had suggested (Carpenter *et al.* 1972), a finding in agreement with Hirayama and Hirano (1970). Thus, given the unrealistically high chlorine concentration during the holding situation in the study by Davis (1983), inhibition of phytoplankton productivity may even be lower in reality.

Survival studies of zooplankton at the power stations mentioned above revealed differences in survival rates depending on geographical location, that is, whether the power stations were located in estuaries or on the open coast (Coughlan and Davis, 1983). In their study, Coughlan and Davis (1983) collected zooplankton from 200-litre volumes of water sampled with a pump sampler designed to minimize sample damage. Their design allowed the plankton to be filtered from the water prior to the water being drawn into the pump. Samples were taken from the power station intakes and outfalls (Coughlan and Fleming, 1978a).

Vital staining techniques to analyze entrainment survival were first tried by Heinle (1976) studying copepods at three power stations in the USA. On collection, a vital stain, Neutral Red, was added to the samples; this stain is only absorbed by live organisms (Fleming and Coughlan, 1978). This technique is considered more reliable and less time-consuming in the field compared with monitoring the motility of sampled zooplankton (which must be assessed immediately upon capture) as a means of establishing mortality (see Dressel *et al.* 1972). Heinle (1976) found poor replication and inconsistencies attributable to stratification of the plankton in the sampled water; numbers of organisms per sample were often below 10. Percentage survivals, measured by those individuals taking up the vital stain, were generally high at both intake and discharge (mostly above 80 per cent) with some examples of reduced survival at the discharge during chlorination. However, quantification was not practical.

The overall conclusion was that sample sizes (hourly one-litre samples) were inadequate, despite the effort and funding such a programme required. As copepods (holoplanktonic Crustacea) are typically abundant in zooplankton samples, adult calanoid copepods were the organisms monitored in the study by Coughlan and Davis (1983), using pumps to obtain larger sample volumes. One-hour post-collection and staining, the samples were preserved and could later be analysed for live (red) and dead (non-red) individuals.

A series of studies on zooplankton survival at French coastal power stations, reported by Khalanski (1978), gave a range of results from increased densities of zooplankton at the discharge (vital-staining tests), no differential survival (post-entrainment incubations), holoplankton mortalities of between 30 and 70 per cent (asynchronous post-entrainment densities), and 100 per cent mortality of sprat eggs and 17 to 61 per cent mortality of sole eggs attributed predominantly to mechanical shock.

Copepods from estuarine environments incurred greater mortality than those from open coastal locations. Mortality rates were considered in relation to seawater quality (poorer in estuaries) and chlorine concentration (Coughlan and Davis, 1983). Although mortality increased greatly with chlorine concentration, compared with phytoplankton it was much lower and under standard operating conditions (ΔT 8-10 °C, <0.2 mg l^{-1} Cl at outfall) the survival of adult calanoid copepods in the zooplankton was above 90 per cent (Figure 6-8; Turnpenny and Coughlan, 2003).

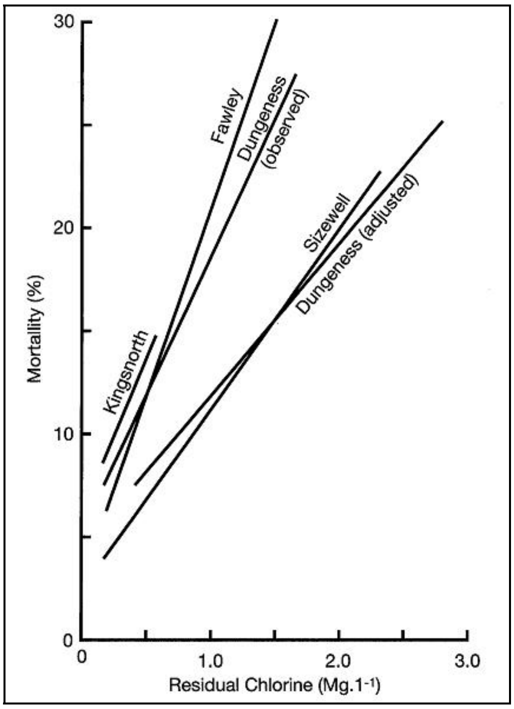


Figure 6-8 Percentage mortality of adult calanoid copepods within one hour of entrainment under various chlorination regimes at four different UK power stations (Source: Coughlan & Davis, 1983)

Laboratory-based studies

The problem with power station-based studies is that the range of taxa available for assessment is largely a matter of chance: while it is feasible to study copepods or phytoplankters in general, as some will invariably be present, no one species can be guaranteed. Further, owing to the sparseness of some species, including commercially important meroplankton, it may be impossible to sample adequate numbers even if they are known to be present at that time and place. Such studies are only able to test the effects of the totality of entrainment, although Coughlan and Davis (1981) were able to vary the biocidal chlorine dosing levels during some of their studies.

Laboratory-studies in theory offer the advantage of controlling the entire experimental entrainment process, including the test animals (guaranteeing results),and stressors. The apparatus should have a practical advantage, as the flow can be passed through a single (full-length) condenser tube, so that a relatively small and manageable flow of water is used. This means that as few as thirty to fifty individuals (eggs, larvae and so on) can be tested per experimental run, with 100 per cent recapture in the collecting vessel, and no significant handling damage.

Early attempts at experimental assessment of entrainment took too simplistic an approach to offer useful interpretation. Poje *et al*. (1981) undertook experiments on estuarine fish and arthropods using a condenser-tube simulator; unfortunately, their apparatus was unable to generate the complex pressure profiles characteristic of power station cooling-water systems.

Kwik and Dunstall (1985) cultured zooplankters in conditions of thermal shock, mimicking entrainment stresses; they generally found that survival was of the order of 90 per cent as long as the test temperature did not exceed 29.5°C, irrespective of ΔT. Obviously, this does not mimic normal operating conditions as only thermal stress was tested.

Schubel *et al.* (1976) exposed eggs and larvae of three fish species from the Chesapeake Bay region (blueback herring, *Alosa aestivalis*; American shad, *Alosa sapidissima*; striped bass, *Morone saxatilis*) to simulated ΔT effects by simply immersing small pots containing the test animals in water baths at different temperatures for between four and 60 minutes, returning them to ambient temperature water-baths for cooling to background temperature (60-300 minutes). They found that ΔTs of 7 and 10°C did not significantly affect hatching success of any species, while a ΔT of 15°C significantly reduced hatching success of both blueback herring and American shad; only striped bass larvae could withstand ΔTs up to 10°C with no significant increase in mortality. A ΔT of 20°C resulted in near total mortality of eggs and larvae of all three species. Despite finding that the fish eggs were apparently more tolerant of ΔT effects than were the larvae, most response patterns were found to be “complicated”. Of course, such buffered thermal impacts do not reflect actual conditions during entrainment faithfully.

Other studies in Europe involved simple tolerance tests of stressors (mainly temperature) in culture. These studies were numerous (being easy to set up), inconclusive and were of limited application to the real entrainment situation (see review by EDF, 1978).

Simulations were conducted in the USA using condenser tubes to assess the mechanical stresses of entrainment (see Cada *et al.* 1981; review by Jinks *et al.* 1981). Large differences were observed between species in their response to pipe and condenser passage and, for most species, short-term mortality associated with passage, estimated to be under five per cent, increased with increasing ΔT and/or pumping rate. However, there was no significant difference in survival between test and control organisms.

An Entrainment Mimic Unit (EMU), a laboratory simulator (Figure 6-9), was developed by Fawley Aquatic Research Laboratories (FARL) in the 1990s to assess entrainment survival (Bamber *et al.* 1994). This apparatus was superior to its predecessors in that it was able to mimic the levels and range of stressors found at power stations; these stressors could be varied individually and applied alone or in combination, allowing the distinction of their effects separately, synergistically or antagonistically.

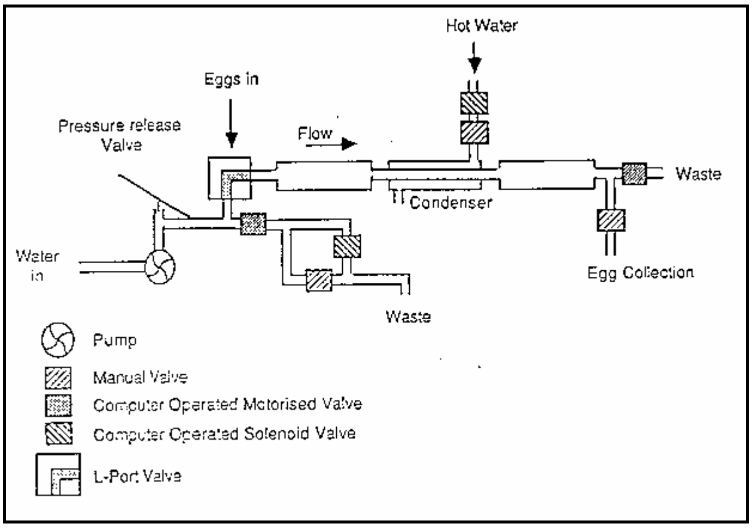


Figure 6-9 Schematic of FARL Entrainment Mimic Unit (EMU) (after Bamber *et al.* 1994)

Computer-controlled solenoid and mechanical valves allowed the replication of a complex pressure profile (normally that of a coastal direct-cooled PWR), while controlling the timing of hot-water delivery to allow a range of ΔT. Effects of antifouling “chlorination” were tested by introducing sodium hypochlorite at a range of test dose levels. The mechanical stresses of entrainment (physical abrasion, collisions and so on) were an inherent feature of the apparatus and as such uncontrollable. Test conditions ranged around typical coastal power station stress-levels of 0.2 ppm of Total Residual Oxidant (TRO) and 10ºC ΔT, normally between zero and one ppm and between 0 and 15ºC.

Tests were conducted on the planktonic stages of a range of species, including meroplanktonic eggs and/or larvae of commercial crustaceans (common shrimp and lobster), commercial fish (sea-bass, Dover sole, turbot), a commercial mollusc (Pacific oyster), fouling species (two barnacles, common mussel) and the holoplanktonic copepod *Acartia tonsa*. A summary of entrainment mortality of these species under “normal” power station levels of stressors is given in Table 6-2, based normally on the condition of specimens 24 hours after entrainment.

Table 6-2 Percentage entrainment mortalities of a range of planktonic species under "normal" power station levels of stressors, interpolated from series of EMU experiments (data from Bamber & Seaby, 1993; 1995a,b; 2004; Bamber *et al*. 1994).

Species	Stage	Mortality at 0.2 ppm TRO and 10ºC ΔT (%)
Crustacea		
Acartia tonsa (Copepoda)	Adults	20
Crangon crangon (common shrimp)	Larvae	25
Homarus gammarus (lobster)	Larvae	8
Elminius modestus (barnacle)	Nauplii	0
Fish		
Dicentrarchus labrax (sea bass)	Eggs	46
Dicentrarchus labrax (sea bass)	Larvae	44
Solea solea (Dover sole)	Eggs	7
Solea solea (Dover sole)	Postlarvae	92
Psetta maxima (turbot)	Eggs	7
Psetta maxima (turbot)	Larvae	70
Mollusca		
Crassostrea gigas (Pacific oyster)	Larvae	95
Mytilus edulis (common mussel)	Larvae	0
Mytilus edulis (common mussel)	Spat	35

What is striking is the wide range of effects between major animal groups, between species and between different life-stages of the same species.

The pressure cycle caused a significant mortality in the copepod *Acartia tonsa*, Pacific oyster larvae, and, combined with the ΔT, inhibited hatching of the flatfish eggs, but affected no other taxa tested.

The physical stresses of entrainment were the only factors to cause significant mortality in turbot larvae (usually loss of yolk-sac) and lobster larvae (usually loss of abdomen), but on no other taxa tested.

The residual from chlorination (TRO) at around 0.2 ppm contributed to the mortality of sole post-larvae, sea bass larvae, mussel spat, Pacific oyster larvae, *Crangon* larvae and *Acartia* adults, but had no significant effect on any other taxon tested.

The thermal stress was resolved into two factors: the ΔT caused significant mortality to sea bass eggs and larvae (with an evident synergism with TRO), and contributed to significant mortalities of flatfish eggs and sole post-larvae; the actual enhanced temperature (ºC) increased the mortality of *Crangon* larvae in response to TRO in an evident synergism.

The results show that most individuals of most taxa (other than flatfish larvae/post-larvae and Pacific oyster larvae) survive entrainment. However, the causes and degree of mortalities are different for different taxa and life-stages, and generalizations, for example for environmental impact assessments, must be undertaken with great care. Interestingly, while the larvae of potential fouling species (barnacle and common mussel) showed 100 per cent survival, individuals were inactive for the first few hours after entrainment: the function of antifouling chlorination is to prevent settlement of these larvae, not necessarily to kill them, and as such clearly works well.

6.1.5 Ecological and commercial significance of impingement and entrainment

Approaches

Various frames of reference can be used to assess impingement and entrainment impacts. In Britain, impacts have normally been assessed relative to particular groups such as commercial fishermen, recreational anglers, or against conservation objectives. In the USA, the California Energy Commission (CEC, 2005) undertook a comprehensive analysis, representing losses in both ecological and economic terms.

The Sizewell protocol

The 1981-82 impingement study at Sizewell provided the first comprehensive UK impact assessment for fish impingement. Analysis of the data showed a total of 73 species of fish recorded over the year. Six of these were commercially valuable species and were caught in quantities of more than a few hundred individuals, their total annual catch amounting to 4.6 tonnes per year. Using population statistics on the expected survival rates and growth rates of these species in the North Sea, it was possible to calculate the likely yield from these fish to the fishing industry, had they not been removed prematurely by the station. This is known as the 'consequential loss'. The required population statistics are routinely collected and published by the International Council for the Exploration of the Seas (ICES) for fish-stock management purposes. The results (Table 6-3) indicated a consequential loss of 66 tonnes per year, less than the catch of a single small trawler. In terms of North Sea fish stocks, this represents a fraction of one per cent (Table 6-3), compared with typical commercial exploitation rates of 10-60 per cent on the same stocks.

Table 6-3 Sizewell A Power Station, 1981-82 Study. Estimated annual loss to the fishery of commercial-sized fish due to CW abstraction (after Turnpenny *et al.* 1988)

Species	Immediate loss (tonnes per year)	Consequential loss (tonnes per year)	% of North Sea stock taken by power station
Plaice	0.03	1.0	0.00072
Sole	0.63	0.9	0.013
Dab	0.41	3.5	0.00034
Cod	1.8	2.8	0.00044
Whiting	1.5	43	0.0087
Herring	0.24	15	0.0017
Total	**4.6**	**66**	

The equivalent adult value (EAV) method

The statistical procedure used in 1983 for Sizewell A (Turnpenny *et al.*,1988) was cumbersome and has since been superceded by the equivalent adult value (EAV) method (Turnpenny, 1988b), which is now widely used in power station impact assessment. The EAV method is also used in the USA (see CEC, 2005). The equivalent adult procedure allows the biological value of fish of different ages to be compared. The concept is based on the fact that a pair of spawning adult fish may produce hundreds or thousands of fertile eggs, but only two of these must survive to sexual maturity for the parents to be replaced and the population to remain stable. Initial mortality due to natural causes (mainly predation) is very high. If, for example, 1,000 eggs are spawned by a pair of three-year-old just-mature parents, after one month there may be, say, 100 surviving larvae; after one year, there may be five remaining and after three years (just mature) there may be two fish left. This relationship is defined by the equivalent adult value curves shown in the graph below. From the curves, it is possible to equate the value of fish caught at any age to that of a just-mature fish. Taking the above example, the one-month-old fish would have an EAV of 2/100 or 0.02, that is, it would take 50 larvae to generate one adult. The numerator here allows for the fact that we are left with two adults to spawn at the end of the cycle. Fish caught at age one year in this example would have an EAV of 2/5 or, 0.4, thus it would take 2.5 one-year old fish to generate one just-mature fish.

Mathematically, the EAV is defined as:

$$EAV = 1 / (S_t \cdot F_a),$$

where F_a is the average lifetime egg production of an adult and S_t is the probability of survival from birth to any future time t. Any fish that survives past maturity has an EAV above one. The EAV can be converted to weight by multiplying the EAV by the average weight of fish in the population at the age of just becoming mature. Figure 6-10 shows EAV curves for commoner British commercial species.

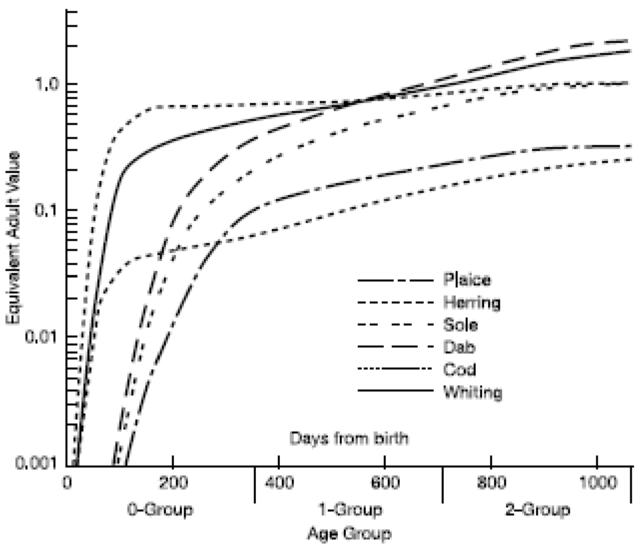


Figure 6-10 Equivalent adult value curves for common UK commercial species (Turnpenny, 1989)

Table 6-4 gives annual EAV tonnages of impinged fish for four UK directly cooled estuarine power stations and compares them with reported commercial landings data from adjacent sea areas. The procedure can be applied to fish of any age, from the freshly fertilised egg onwards, and is equally applicable to entrainment data, therefore (see Section 6.1.3: *Entrainment at UK inland stations*).

A problem with applying the EAV method to early-stage fry is that these fish have r-selected reproductive strategies (Reed *et al.* 1988), species of this type tending to produce comparatively large numbers of young to maximise use of available habitat following, for example, catastrophic events such as drought, flood washout or pollution. Mortality in these stages is naturally high. Thus, it is intuitively unreasonable to assume that a given percentage mortality in fry numbers will lead to a proportionate reduction in adult stock: this will only occur when fish stock densities are low enough, or juvenile mortalities high enough, to prevent carrying capacity being reached (Van Winkle, 1977) except where migration is possible. A counterargument to this is that the profuse larval production represents the population's safety margin to cope with stochastic environmental change and that increasing mortality erodes the natural elasticity of the population (Reed *et al.* 1988).

The EAV technique should be regarded as an accounting procedure rather than a biological model: it is used to enable comparisons of fish of different ages on a like-for-like basis. Its main flaw as a biological model is that it does not take account of variable population growth associated with the density of the fish population, where it will normally be the case that overcrowded habitats will lead to higher mortality rates and lower growth rates due to competition for resources, increased disease risk and so on. This implies a degree of self-regulation of population size or biological 'compensation' (Horst, 1977; CEC,2005). Consequently, use of the EAV method without taking account of these so-called 'density-dependent' effects will tend to overestimate losses due to entrainment and impingement, giving a 'worst case' scenario. Unfortunately these population compensatory effects, which are intuitively plausible, are almost impossible to demonstrate, which led Nisbet *et al.* (1996: in CEC, 2005) to conclude, "*Optimistic outcomes (of compensation) all appear to demand mechanisms which have not been proved in any marine fish anywhere*." In further considering this question, US EPA and California Energy Commission have adopted the view that "*compensation does not reduce impacts from entrainment and impingement on adult populations*" (CEC, 2005, Appendix C).

Combining EAV from entrainment with adult losses from impingement provides an overall estimate of adult mortality caused by the cooling system.

Table 6-4 Equivalent adult tonnages of key commercial species impinged at four UK estuarine power stations, compared with reported England & Wales commercial landings from adjacent sea areas (Turnpenny, 1988b).

Species	Age at 50% maturity	Mean weight at age (kg)	Equivalent adult catch (tonnes per year)			
			Heysham I	Hinkley B	Fawley	Kingsnorth
Plaice	5	0.465	0.411	0.014	0.012	0.656
Sole	3	0.229	0.033	0.306	0.040	1.10
Dab	2	0.1	0.151	0.297	0.004	0.002
Cod	4	4.36	0.100	5.61	0	0.043
Whiting	2	0.178	0.689	10.8	0.078	4.44
Herring	2	0.126	1.91	0.333	5.4	4.01
Total			3.29	17.4	5.53	10.3
Total 1986 landings			9,270	5,485	3,172	11,655
ICES Sea Area			VIIa	VIIf	VIId	IVc

Comparison with fishery discards

Another useful comparison can be made with quantities of fish discarded by the fishing industry. Vessels at sea target particular saleable species and fish that are above the minimum statutory landing size (which varies from species to species). The fishing gears used and areas fished are designed to target these as accurately as possible but there are inevitably incidental catches of undersized fish and less saleable species that are discarded back to sea (known as 'bycatch'). The bulk of these do not survive return and act as food for other fish, seabirds and marine mammals. A recent Norwegian study estimated the annual fish discard rate in the North Sea to be 146,000 tonnes of roundfish and 148,000 tonnes of flatfish (Camphuysen and Garthe, 2000).

Research at Sizewell A Power Station showed that fish catches by the station amounted to 0.76 million fish per annum, compared with an annual bycatch of 406 million fish per annum in the Wash and German North Sea shrimp fisheries (Turnpenny *et al.* 1988). A similar study at Heysham Power Station compared the screen-catches of 8,000 plaice and 16,000 sole juveniles per annum with the bycatch of these species in the local Morecambe bay shrimp fisheries, estimated at 1.7-5.8 million plaice and 0.14-3.9 million sole (Turnpenny, 1988b).

Habitat production foregone (HPF)

While the previous three methods primarily consider impingement and entrainment losses within a commercial fishing context, this and the next method are of more general ecological and conservation interest. The habitat production foregone (HPF) concept (also known as *equivalent area of lost production*, EALP: Turnpenny, 2002) has been used to assess entrainment and impingement losses at California's directly cooled stations (CEC, 2005) and allows quantities of fish removed by power stations to be equated to the equivalent area of marine habitat being taken out of production. This is particularly useful when considering ecological requirements in compensation for residual impacts once other mitigation measures have been applied. Table 6-5 gives indicative figures for Californian stations. It would be misleading to assume that the size of the impact is related to the size of the plant, as Figure 6-11 reveals.

Table 6-5 California Energy Commission figures for habitat areas that would be required to replace entrainment losses at Californian power plants (CEC, 2005). Plant CW flow shown for comparison.

Plant	Flow rate (cumecs)	Area of replacement habitat (Ha)
Diablo Canyon	127	120-240
Morro Bay	33	93-307
Moss Landing	61	460
Potrero	11.3	357
San Onofre	52	61
South Bay	30	406

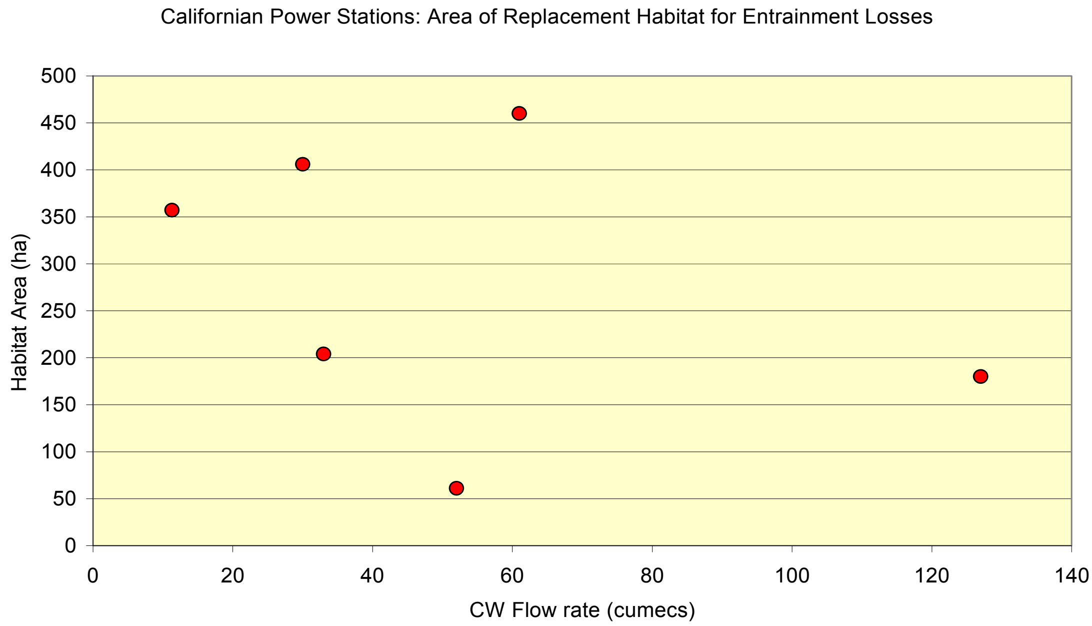


Figure 6-11 Area of replacement habitat for entrainment losses against flow rate (plotted from data in Table 6-5)

Habitat production figures can be obtained from the literature, although not always for the geographic area in question. For example, work carried out on saltmarsh habitat by Nixon and Oviatt (1973) at Bissel Cove, New England, USA, estimated net export production of the salt-marsh at 250 kJ $m^{-1}y^{-1}$. Assuming this went into fish production with an energetic value of 4.5 kJ g^{-1}, each square metre of saltmarsh would support 56 g y^{-1} in fish production.

As an example closer to home, fish production for the North Sea as a whole has been estimated at 2,500 kg km^{-2} y^{-1}, while estuarine production estimated for the Forth was 4,300 kg $km^{-2}y^{-1}$ (Elliott and Taylor, 1989). Equivalent figures have been obtained for other water bodies. These figures can be applied directly to power station fish catch estimates. For example, 43 tonnes of fish per year impinged at Sizewell (Figure 6-5) equates to an equivalent area of lost North Sea production of 17.2 km^2 on this basis.

A similar approach was used by Turnpenny (2002) in the context of Fawley power station (Hampshire), where it was estimated that the annual catch of impinged fish at a CW flow of 32 $m^3 s^{-1}$, expressed in equivalent adult terms, was 424 kg y^{-1}. Using production figures for the Forth Estuary given by Elliott and Taylor (1989), this is the equivalent to lost production of 9.9 ha. Comparison was made with the Forth Estuary owing to lack of more local data on estuarine production; this was based on the whole area of the Forth Estuary, including sub-tidal areas.

Lost food source for piscivores

This method is particularly appropriate when considering indirect impacts of a power station development on species of conservation interest. Previous cases have investigated impingement and entrainment losses in terms of the annual dietary requirements of grey seal (*Halichoerus grypus*) and seabirds. The grey seal (listed in Annex II of the Habitats Directive), for example, which feeds on available inshore fish, cephalopods and crustaceans, consumes an estimated 7.5-12.5 kg daily (Bonner, 1982), equivalent to 2.7-4.6 tonnes per year. On this basis, loss due to impingement at the power station equates to a percentage or multiples of the annual diet of a grey seal.

The dietary consumption of piscivorous seabirds has been estimated by Camphuyson *et al. (*in Turnpenny, 2002) who investigated the number of seabirds that would be supported by fish discarded from fishing vessels. Such discards comprise undersized fish and those of non-target species that are thrown back to sea. Their figures provide a basis for assessing the number of seabirds supported by fish removed by a power station. They are based on a hypothetical 1,000 g seabird, with a daily energy intake (3x basal metabolic rate) of around three times 600 kJ per day, or 6.57 x 10^5 kJ y^{-1}. The calorific value of fish was estimated at 5 kJ g^{-1} for roundfish, 4 kJ g^{-1} for flatfish. Using these figures, fish removed by the power station can be equated to the "lost food" of a number of 1,000 g seabird equivalents. Taking the average value of 4.5 kJ g^{-1}, the 0.4 tonnes of fish removed by Fawley Power Station was estimated to equate to 1.8 x 106 kJ y^{-1}, or the dietary requirements of nearly three 1,000 g seabirds (Turnpenny, 2002).

6.1.6 Design best practice and mitigation techniques

Cooling water intake location

Earlier sections of the report dealt with engineering and plant performance factors, such as recirculation and air entrainment that influence CW intake location. Here, we look at how intake location should take account of biological factors in the local area. Biologically-informed planning not only helps to reduce impacts upon fish, crustaceans and other valuable biota; it also minimises risk of periodic inundations of fish and weed that can otherwise lead to serious operational problems and potentially large revenue losses. Selecting an intake location is an iterative process that takes account of all the above factors. Below we outline some biological considerations.

Fish-screening and fish-return techniques available today greatly reduce the risk of fish losses to impingement, whereas entrainment of early life-stages is harder to control. Therefore, biological considerations should focus more on avoiding entrainment risk than impingement.

A key reference that should be used alongside this section is the Environment Agency's Best Practice Guide for intake and outfall fish screening (Turnpenny and O'Keeffe, 2005). The Guide contains a great deal of information on environmental mitigation techniques for intakes, as well as background data.

Locality of power station

Generally, it is desirable to avoid construction in areas of notable aquatic habitat conservation or ecological value. These might include, for example, Natura 2000 sites[24], important fish spawning and nursery grounds, ecologically sensitive habitats, economically important shellfisheries and fish migration routes. Owing to the UK Government's rules on site selection for new nuclear build, there is little scope for moving the base locations of proposed developments and therefore siting questions must revolve around whether the best decisions are being made within the geographic planning constraints and whether the residual impacts after mitigation are acceptable or compensatable. Given that CW inlet and discharge tunnels can practically extend to a few kilometres in length, and that the proposed sites for new nuclear power stations

[24] Natura 2000 sites are sites identified as of Community Importance under Habitats Directive 92/43/EEC or classified as Special Protection Areas (SPAs) under Birds Directive 79/409/EEC

have sufficient land-area to build cooling towers etc, there nevertheless remains scope for minimising or avoiding impacts upon important habitats.

It should be noted that an Appraisal of Sustainability (incorporating the requirements of the Strategic Environmental Assessment (SEA) Directive 2001/42/EC) and a Habitats Regulations Assessment are being completed alongside development of the Nuclear National Policy Statement (NPS). These plan-level assessments inform the NPS and should address some of the issues above.

Shoreline, mid-channel or offshore options

Key considerations here will be the relative importance of different habitats represented in these zones.

In any coastal or estuarine situation, drawing a transect from the shoreline into deeper water will reveal strong biological zonation. Intertidal and saltmarsh areas are often the most highly productive, and are also important habitat for juvenile fish. Abstraction from these areas will increase the risk of drawing in juveniles. An example of this is seen at Fawley Power Station, Hampshire, which has an onshore intake connected to the deep shipping channel of Southampton water via a 500 m dredged channel. The adjacent saltmarshes drain into dredged channel as the tide falls, concentrating fish in the CW stream and increasing entrapment risk (Turnpenny and Utting, 1981). Had it been recognised as an issue at the time of construction, such an arrangement might have been avoided by locating the intake offshore via a pipeline, or by sheet piling the intake channel edges to ensure that the saltmarshes drained away from the dredged intake channel (Figure 6-12). A similar risk may also arise with an intake that opens near the low water mark, where fish in the intertidal tend to concentrate as the tide ebbs.

Figure 6-12 Fawley Power Station CW intake at low tide, showing the onshore CW inlet channel and bordering saltmarsh areas

Cefas and its predecessors in MAFF DFR have for many years conducted Young Fish Surveys around the coasts of England and Wales (Riley *et al.* 1986; Rogers *et al.* 1998). These used two standard scientific fishing techniques, a two-metre beam trawl and Riley 1.5-m push-net to sample coastal habitats starting at the surf zone and extending into deeper waters to around 20 m. Data show zonation, particularly of flatfish species, zero-group concentrations being highest in the surf zone, giving way to 1+ and 2+ fish in deeper water (Figure 6-13). Data from these surveys for the east coast were a key factor in the choice of CW intake location for Sizewell B, where intake was pushed 600 m offshore (compared to 300 m offshore for the A station) to reduce the risk of drawing in weakly swimming zero-group fish (Turnpenny and Taylor, 2000).

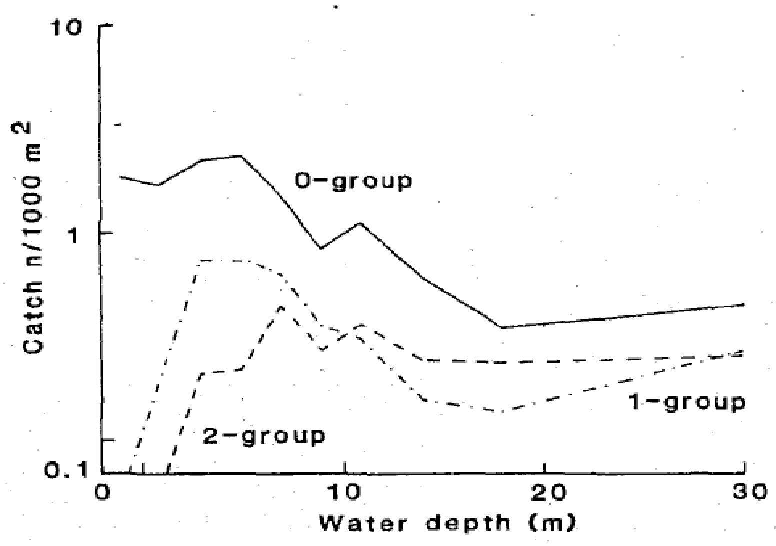


Figure 6-13 Cefas Young Fish Survey data showing zonation of sole (*Solea solea*) relative to water depth

Siting intakes within narrow estuaries brings a different problem. While locating the CW intake offshore reduces the risk of juvenile fish entrapment, it may increase the risk for any migratory fish such as salmonid smolts that migrate in mid-channel.

Given the potential for entrainment of early life-stages, intake siting should take particular account of spawning and nursery areas. Generally these locations are well known and mapped and should be shown on any GIS maps developed for the project. More important commercial ones are shown on UK DMAP[25] and can be found for example on UKOOA Fisheries sensitivity maps[26] and other marine GIS resources. Such sources may not include spawning grounds that are locally rather than nationally important. For example, a herring spawning ground at Beggar's Reach in Milford Haven (South Wales), which supports a spawning stock of 10-20 million adults, is of no significance to UK herring landings but may be an important ecological feature and contributes occasionally to the local inshore fishing industry (Clarke and King,1985).

When positioning of an intake occurs within a few kilometres of an inshore spawning ground, current patterns need to be taken into account. This applies to estuaries and the open coast, as tidal excursions will carry early life-stages this sort of distance on a single tide. Dispersion of larval herring from the River Blackwater herring stock, which is a Thames spring-spawning sub-stock, was investigated by the CEGB as part of the 1980s Bradwell B programme (Henderson and Cartwright, 1980). A slightly different modelling approach was later applied to the same stock by Fox and Aldridge (2000). Herring are benthic spawners, with adhesive eggs that attach to gravel beds until hatching. Once released into the water column, the larval herring disperse by a process of diffusion and current-driven drift. Samples were collected at various points down-

[25] Available as a download from http://www.bodc.ac.uk/products/bodc_products/ukdmap/ (accessed 17/02/09)
[26] Available as a download from http://www.cefas.co.uk/publications/miscellaneous-publications/fisheries-sensitivity-maps.aspx (accessed 17/02/09)

tide towards the proposed intake location, which was 12 km from the Eagle Bank spawning ground. Both studies reached similar estimates of between 18 and 25 per cent of larvae being washed past the section of channel in which the intake was to be located over the 70 days following hatching (though only a proportion of these would be entrained). This type of approach is more useful for benthic spawning species as the spawning areas are more discrete than those of pelagic spawners such as sole, plaice or bass. Other benthic spawners include, for example, shads (*Alosa* spp.) and sand smelt (*Atherina boyerii*).

Recent investigations have been undertaken as part of the environmental assessment process for the planned Pembroke CCGT power station, whose CW intake will be located about 10 km from the Beggar's Reach herring spawning ground mentioned above. Ichthyoplankton surveys close to the intake indicate that numbers of larval and post-larval herring in the CW volume required to supply the station during the post-hatching period would amount to no more than around 0.1 per cent of Milford Haven stock when expressed as adult equivalents.

The biological consequences of choice of intake opening depth have seldom been considered in past intake designs. As discussed in Section 5, the engineering preference is for deeper openings to avoid warmer surface layers and vortex formation. Fortuitously this is probably the better position for avoiding fish entrapment, as the bulk of fish drawn in are commonly pelagic species that favour the mid-to-upper water column. Bottom openings, conversely, will favour demersal and benthic species and the best arrangement may in most cases be to have openings a meter or so off the bed. This aspect needs to be considered on a site-specific basis according to conservation priorities.

Intake siting should also take account of the risk of operationally harmful inundations of biota. Little can be done to avoid exposure to mass inundations of jellyfish or sprat shoals in areas where these reach nuisance levels, as they are ubiquitous on these occasions. The emphasis in these cases should be on designing screening or deflection systems that can cope. Seaweed inundations, on the other hand, usually occur following storms, particularly after autumn dieback. Hydrographic factors then cause the material to accumulate in backwater areas or depressions in the seabed. Avoiding these areas when locating the intake can prevent later problems.

Biota exclusion and deflection techniques

Exclusion of fish, shrimps, weed and other organisms at the point of entry from the source water is greatly preferable to screening and returning them once they have entered the cooling system. Positive exclusion by fine mechanical screens is one approach but as discussed in Section 5.5 is unlikely to be considered for new nuclear sites. Other techniques depend on physical processes such as air bubble curtains to divert biota or upon behavioural stimuli to drive fish away from inlets.

Air bubble curtains

Bubble curtains are formed when a porous or perforated pipe is affixed to the seabed and fed with compressed air. The rising curtain of bubbles so formed has a variety of useful properties in CW intake applications, including the ability to deflect organisms and trash and to reduce the risk of surface oil entering and coating the heat exchangers following a spill.

The mechanism by which this occurs is illustrated in Figure 6-14. In the case of organisms in the water column, as they are drawn towards the bubble curtain, they can become entrained into the vertical convection current generated by the rising bubble plume and brought to the surface. This process is probably also partly due to attachment by surface tension of bubbles to the organism, increasing buoyancy. Once at the surface, a surface current generated by the upwelling plume repels organisms away from the bubble curtain. By laying the bubble curtain diagonally with respect to tidal flow direction, current vectors perpendicular to the barrier line deflect organisms to one side. Figure 6-15 shows the concept in plan view for an offshore intake.

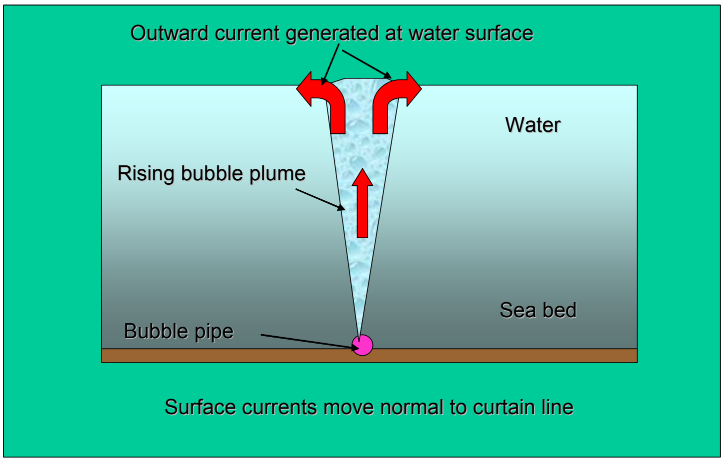


Figure 6-14 Schematic showing rising bubble plume and surface currents normal to the barrier line in a bubble curtain

A bubble curtain was installed during the 1990s across the combined CW intake entrances of Heysham A & B nuclear stations in Heysham Harbour (Turnpenny, 1993) and is still in use. The main intention was to reduce fish ingress at a time prior to the development of more modern methods (see below). The curtain extended across a diagonal line such that fish would be diverted away from the inlet and back into the harbour. Comparison of drum screen catches for alternating on-off periods showed that fish catch was reduced by about one-third. An unexpected benefit was that shrimp catches were reduced by around two-thirds, presumably as a result of the purely physical processes described above.

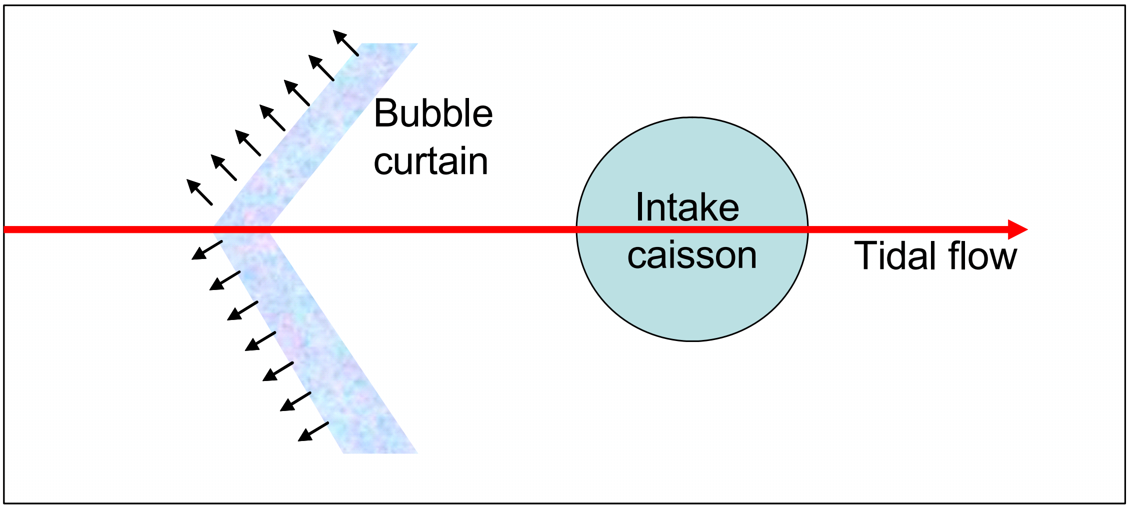


Figure 6-15 Schematic showing bubble curtain deflection concept for an offshore intake

While newer methods can offer better levels of protection for fish, the merits of the bubble intake caisson operating in a tidal flow curtain for deflection of invertebrates and against oil ingress in port and harbour locations should not be ignored. The concept drawing in Figure 6-15 is based on a proposed scheme by Fawley Aquatic Research (now Jacobs Engineering) to reduce jellyfish ingress at Hunterston B nuclear station. The idea is to raise jellyfish in the water column by air becoming trapped beneath the umbrella, taking them over the top of the submerged intake structure, the chevron shape of the curtain at the same time pushing them outwards, away from the intake. It has not been implemented to date.

Bubble curtains have also been used to reduce siltation risk at some power stations, where accretion can partially block openings and cause intake velocities to exceed licensed conditions. An experimental curtain was used to dislodge silt depots around Barking power station on the Thames Tideway but was found to be difficult to maintain. Another is used across the CW intake channel of Fawley power station to reduce sedimentation across the inlet at times when the plant is not drawing CW flow.

Velocity control

Active or behavioural deflection of fish first requires velocities at the intake entrance (often known as "approach velocities") to be low enough for fish to avoid. Design criteria for fish conserving intakes therefore rely on good knowledge of the swimming performance of different fish species and life-stages. As fish are poikilotherms (body temperature is determined by external water temperature), water temperature has a strong influence on metabolic rates and therefore swimming performance.

At one time, relatively crude swimming-speed criteria based on non-indigenous fish species were used for this purpose. More recently the power and water industries, as well as the Environment Agency, have seen the need to develop a database for UK inland and saltwater species. Relevant information is summarised in the Best Practice Guide (Turnpenny and O'Keeffe, 2005). For most power plant intake purposes a design fish-escape velocity of 0.3 ms^{-1} will be suitable and meet best practice requirements. Where a different value might be preferable, the guide should be consulted.

Two further design issues relating to velocity are relevant to offshore intake structures. One concerns elimination of vertical velocity components, which fish are ill-equipped to resist. The problem can be overcome by fitting a "velocity cap" over the intake to ensure that water enters horizontally. Care should be taken to ensure that the aspect ratio (height:length) of the entrance meets the criteria presented by Schuler and Larson (1975) (Figure 6-16), which ensure that vertical components are eliminated. Velocity caps are a powerful form of mitigation that can reduce entrapment by up to 90 per cent, and are already used in some newer UK offshore intake designs (such as Sizewell B: Turnpenny and Taylor, 2000).

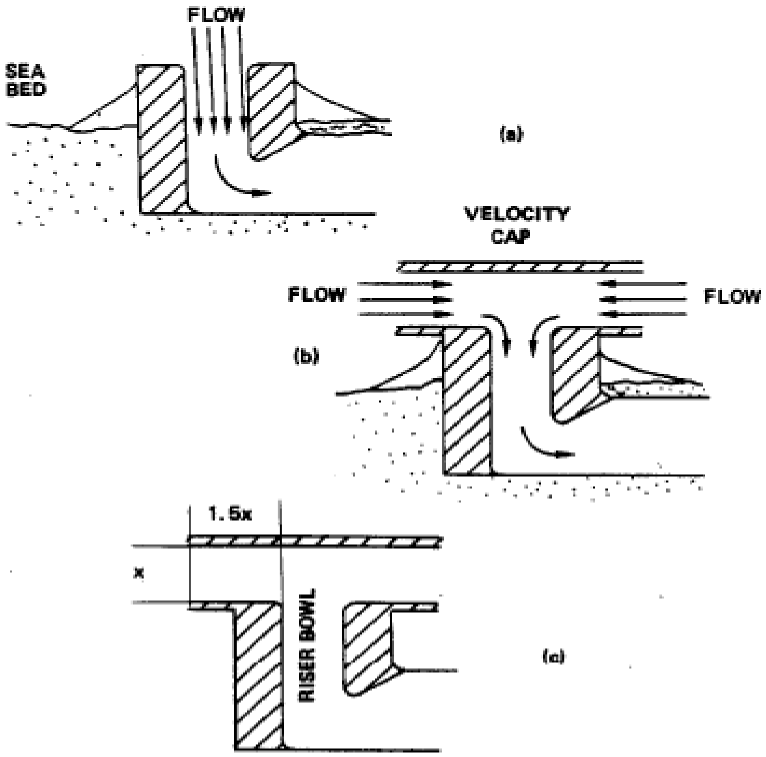


Figure 6-16 Velocity cap: (a) section of uncapped intake showing vertical draw-down pattern, (b) section of capped intake showing horizontal flow pattern, (c) as (b) but showing critical relationship between vertical opening [x] and length of horizontal entrance [1.5x] for fish reactions (after Schuler and Larson, 1975)

The second issue concerns the orientation of intake openings in relation to tidal or river flow. Many UK stations with offshore intakes can take water from 360^{o} around the circumference of the structure. Hydraulic modelling studies have shown that resulting velocities on the upstream side of the intake approximate the sum of the intake velocity (calculated as if there were no crossflow) and the approaching tidal stream velocity (Turnpenny, 1988a). Whereas Sizewell B was designed with a nominal entrance velocity of 0.5 ms^{-1} under static water conditions, the actual approach velocity can be several times this value in mid-flood and mid-ebb periods, leading to higher fish impingement rates at these times. The Best Practice Guide (Turnpenny and O'Keeffe, 2005) presents a conceptual design (Figure 6-17), the low velocity side-entry (LVSE) intake that appears from initial modelling tests to overcome the problem and maintain constant velocities around the tidal cycle. As yet, this design has not been built and requires further model testing to develop the detailed design and ensure its suitability.

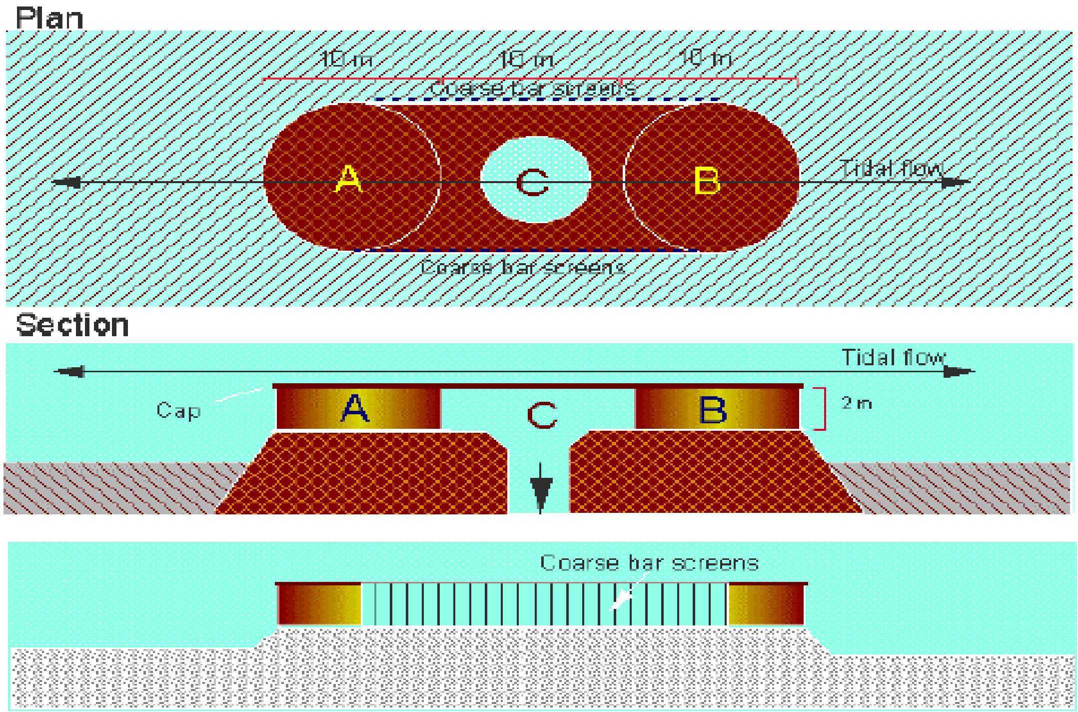


Figure 6-17 Concept design for a low-velocity, side-entry offshore intake structure with velocity cap, based on physical and hydraulic model tests carried out at Fawley Aquatic Research Laboratories (Turnpenny, 1988a and unpublished). A and B are the cylindrical caissons. Water enters through C.

Acoustic fish deterrents and strobe lights

A recent innovation to reduce fish entry into CW intakes is the acoustic fish deterrent (AFD) system. AFDs were originally investigated by the CEGB for Severn Barrage tidal power generation, where the need to exclude fish from hydropower turbines passing up to 200 m^3s^{-1} was considered to preclude conventional physical screening methods. When the Severn Barrage project of the 1990s was dropped, the nuclear industry saw the benefits of pursuing AFD development and early trials were carried out at Hinkley Point (Turnpenny, Thatcher and Wood, 1994) and Hartlepool (Turnpenny *et al.* 1995) nuclear stations. Hinkley Point trials showed that fish impingement rates could be affected by the AFD's sound field, but catches increased rather than decreased! The AFD system used at Hinkley was made up of off-the-shelf military hardware and was not ideally suited to generating frequencies audible to fish. The reason for increased fish catch appeared to be the sound field causing fish to concentrate at the depths from which water was abstracted. For the second series of trials, conducted at Hartlepool, more suitable acoustic transducers ("sound projectors") were developed. These, along with better use of acoustic modelling software to optimise the sound field, led to the successful deflection of a large proportion of fish. Following privatisation of the CEGB and the nuclear industry, independent private sector development continued the refinement of AFDs, there now being a variety of AFD systems suited to different uses.

AFD systems have become commonplace at large abstractions where physical fish screening (as opposed to trash screening) is problematic. They have been installed at large thermal stations such as Doel nuclear station (Belgium), Fawley (oil-fired, Southampton Water), Shoreham (CCGT, West Sussex), Great Yarmouth (CCGT, Norfolk), Marchwood (CCGT, Southampton Water) and Staythorpe (CCGT, R. Trent) and systems are scheduled for installation at other new stations. A system was recently installed at Lambton Generating Station, a large direct-cooled plant on St Clair River in Canada.

The fish deflection performance has been investigated at a number of installations. Testing usually involves comparing screen catches for a sequence of AFD “off” versus “on” days. The cases most representative of the latest technology combined with use of acoustic modelling techniques are Doel (

Table 6-6) and Lambton. The effectiveness of the system depends mainly on the hearing sensitivity of individual species, although other factors such a swimming ability must be taken into account. Most fish possess a swimbladder, which increases sensitivity to sound pressure. Such species (which include many demersal species, including most members of the cod family, bass and other ‘roundfish’) can be deflected with sound. AFD deflection efficiencies are usually between 50-70 per cent for these species. A smaller group of species, known as ‘hearing specialists’, have anatomical adaptations that increase sensitivity to sound. Examples include pelagic fish such as herring and sprat, but also members of the carp family and catfishes. Deflection efficiencies are correspondingly higher, around 80-95 per cent, for these species. A third group comprises flatfishes and other benthic species with reduced or no swimbladder function. Efficiencies are considerably lower for these species. In theory, installing more sound projectors and increasing sound levels will increase the efficiency for all species but the law of diminishing returns applies, and it is normally more cost-effective to add a second type of stimulus, such as strobe lights, and/or to combine AFDs with a fish recovery and return system.

AFD systems that have been used most successfully at power stations deliver sound signals in the 10 to 3,000 Hz hearing band that is audible to most fish[27]. Systems comprise a single signal generator, designed to provide a suite of signals selected according to species, a bank of audio power amplifiers, and a sound projector connected to each amplifier (Figure 6-18). Each line can also be provided with a diagnostics unit, which allows the performance status of all elements of the system to be monitored remotely, for example in the plant control room or via a website.

Sound projectors (Figure 6-19) are installed in a matrix or array across intake entrances, creating a localised, repellent sound field. Normally these are attached to vertical rails incorporating a winch mechanism that enables them to be drawn to the surface for maintenance (at least annually). Figure 6-19 shows the substantial sound projector support structure used at the Canadian Lambton station, designed to ensure that the sound field extends to a point where velocities are low enough for fish to escape. An important consideration for nuclear stations under construction, especially those with offshore CW intakes, is to ensure that suitable cable ducts and cable runs are provided between the onshore station buildings and the intake structure. As cables are difficult to replace, it is also good practice to provide additional spare cabling to allow for future expansion of the AFD system, if necessary, or to replace failed cables.

[27] Ultrasound (>100 kHz) systems have been used with some success in the USA for certain species with ultrasound sensitivity (clupeid family) (Turnpenny and O’Keeffe, 2005) but are not effective for other species and would not therefore be suitable for new UK nuclear build.

Table 6-6 Deflection efficiencies reported for the acoustic fish deflection system at Doel nuclear station (Maes *et al*. 2004)

Fish species	Deflection efficiency ('on' versus 'off')	Statistical significance
Herring (*Clupea harengus*)	95%	P<0.001
Sprat (*Spratus sprattus*)	88%	P<0.001
Smelt (*Osmerus eperlanus*)	64%	P=0.004
Bass (*Dicentrarchus labrax*)	76%	P<0.001
Flounder (*Platichthys flesus*)	46%	P=.0.028
Gobies (*Pomatoschistus* spp.)	50%	P>0.05 (NS)

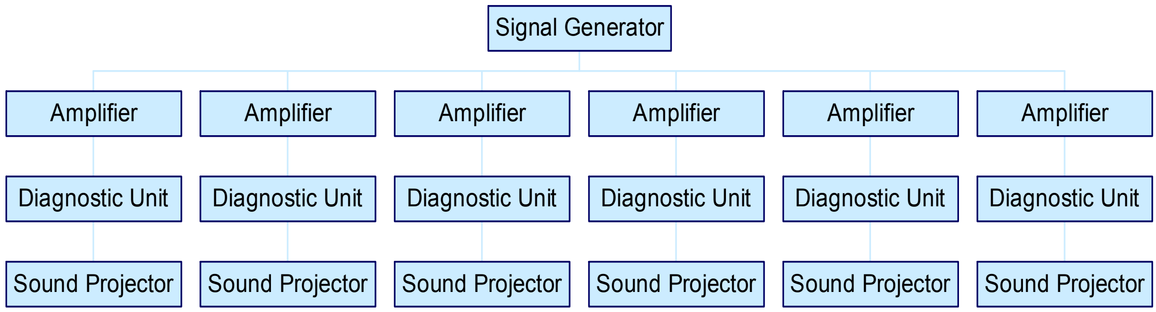


Figure 6-18 Schematic layout of a sound projector array (SPA) acoustic fish deterrent system with six sound projectors. Large systems may use up to 60 sound projectors, each with a dedicated amplifier and diagnostics unit.

Figure 6-19 Example of an acoustic fish deterrent support structure constructed across a shoreline CW intake at the Lambton Generating Station, St Clairs River, Canada. The vertical beams each act as rails, which allow sound projectors to be raised and lowered for maintenance. The positioning of the structure ahead of the intake opening ensures that the fish repulsion zone is in an area of low water velocities from which the fish can escape. The inset shows an individual sound projector. (Main photo courtesy Paul Patrick, Kinectrics).

Strobe lights have been used for many years to deter fish, but have been limited by the high voltage requirements of traditional xenon strobes (kilovolts) and their restricted bulb life. The recent development of low-voltage (under 12 V) strobes using high-intensity light-emitting diodes (LEDs) has overcome these problems and the abstraction licence for the proposed Pembroke CCGT requires strobes to be added to the AFD system on an experimental basis. Strobes appear to be one of the few effective means of deterring eels, although they are also expected to augment the AFD deflection efficiency for other fish species. More information on strobe and AFD systems can be found in the Best Practice Guide (Turnpenny and O'Keeffe, 2005).

Biota recovery and return techniques

Fish recovery and return (FRR) systems derive from standard band and drum screen designs but incorporate modifications that reduce the risk of fish injury (see Section 6.1.3 for a description of stressors). Key features are:

- replacement of standard trash elevators with water-retaining fish buckets;
- continuous screen rotation so that fish are not impinged against the screens for long periods before removal;
- use of low-pressure backwash sprays for fish removal (usually followed by high-pressure sprays to clear more persistent fouling);
- ensuring backwash launders are smooth, free from potential snags and flushed with copious quantities of water (Figure 6-20);
- a fish return pipe or launder to put fish back to the source water body at a point where they are unlikely to be returned to the intake point. The fish return line should enter the water below the lowest astronomical tide mark.

Figure 6-20 Example of fish return launders. Launders should be covered to reduce predation risk. Larger radius (3 m) swept bends reduce the risk of debris and fish becoming caught in bends.

Research has shown that the design of the fish buckets is critical, if fish are not to be flushed out of it by the shearing effect of water being pumped through the screen mesh (Figure 6-21a). The problem is avoided by using buckets of a suitable hydraulic design (Figure 6-221b), recently fitted, for example, at the new Marchwood CCGT station.

A new issue in the design of fish buckets is the need to handle large, sinuous species such as adult eels and lampreys. Survival rates of eels in fish return studies have generally been low (see Clough *et al.* 2003). The main reason is that eels writhe and fall out of the buckets, probably many times before being removed by the screens. As a result they become exhausted and often have multiple wounds and sores by the time they are removed.

The same may be true of lampreys, although observations are fewer. Fish buckets typically have an opening width of 60 mm (Figure 6-22) which may be too small to handle these species safely. Given the recent EC Eel Regulation and the conservation status of lampreys under the Habitats Directive, there is an urgent need to investigate this issue and develop suitable design criteria.

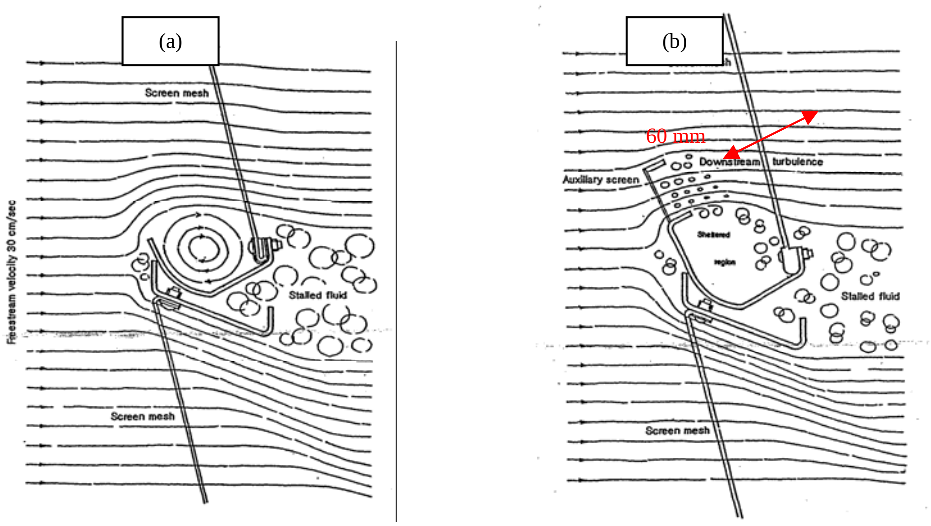


Figure 6-21 Streamlines around a travelling screen fish bucket (a) traditional, showing rotational flow and (b) modified to stall flow within bucket (Fletcher *et al.* 1988). Width of bucket opening is typically 60 mm in standard designs.

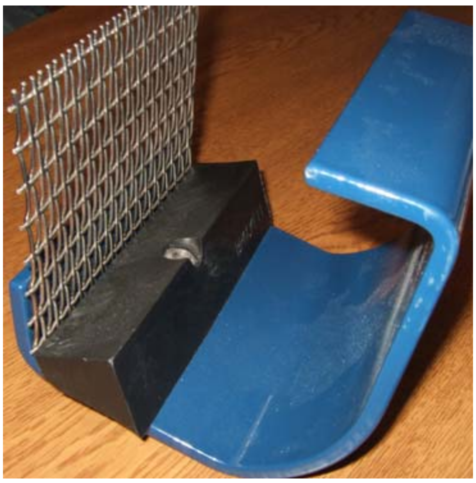


Figure 6-22 Example of a fish bucket profile. The mesh to the left represents part of one of the main screen panels (Eimco Water Technologies).

Biocides used for biofouling control within the CW system should ideally be injected downstream of the screenwell to avoid toxic stress to the fish being returned. While Turnpenny (1992) concluded that brief exposure to TRO may be tolerated by fish, exposure to sublethal effects of TRO may disorientate fish and increase the risk of predation once they are released back to the wild. The Best Practice Guide (Turnpenny and O'Keeffe, 2005) therefore recommends that exposure be avoided, although screenwells are often chlorinated. This recommendation has yet to meet with a satisfactory response from developers, who are concerned about biofouling accreting within the screenwells and potentially blocking screens, a problem which may need to be addressed with intermittent shock dosing or use of antifouling coatings.

The Best Practice Guide gives the following summary of design criteria for FRR systems, which should be applied as far as possible at new nuclear stations:

- The design of the fish buckets should be optimized for fish handling.
- Screen meshes should be smooth and fish-friendly, constructed from woven stainless steel or plastic mesh.
- Mesh size should be 6 mm or less.
- Low-pressure backwash sprays (under one bar) should be used for fish removal; higher pressure jets may be used at a later point in the cycle to wash off debris.
- The geometry of the collecting hoppers should ensure that fish washed off the screens cannot fall back into the screenwell.
- Biocides should be applied only downstream of the screens unless it can be shown that the toxic risk is negligible.
- Fish return gullies should be smooth, with any joints properly grouted and finished. They should have a minimum of 0.3 m diameter with at least 0.5 m diameter or larger for the main return channel run (above 30 m).
- Fish return lines should be covered to prevent bird predation and algal growth, with access for cleaning.
- Swept bends of radius greater than three metres should be used.
- A dedicated fish return line should be provided.
- A continuous wash-water supply should be provided to ensure sufficient depth to keep fish immersed and moving along the return line.

Putting these guidelines into practice at recent new-build stations has already revealed further details that should be specified, and other requirements may come to light as experience is gained. The following additional design criteria should be considered:

- In order to minimise the fish handling time the screens should be rotated at a constant speed of at least 1.5 m per minute.
- Any changes in slope of launders should use a minimum 3-m swept bend radius for vertical bends to avoid flow separation from the launder bed.
- The fall on launder sections feeding into horizontal bends should be restricted, as accelerating flow may cause standing waves and overtopping in the bends. This may be accomplished, for example, by restricting the fall on launder sections upstream of horizontal bend to 1 maximum of 1:50.

- Turbulence should be minimised to reduce the risks of fish exhaustion and injury. It is recommended that energy dissipation throughout the system should be kept at or below 100 Wm^{-3}. This particularly applies to any fish sampling or holding facility that may be incorporated for fish impingement monitoring purposes. The method of calculating energy dissipation is described in the Environment Agency Fish Pass Manual (Armstrong *et al.*, 2004).

- While most FRR systems allow the high-pressure backwash (primarily intended to remove material sticking to the screen) to discharge to a trash basket, provision should also be made to allow discharge of backwashings via the fish return launders. This facility can be used in cases where significant quantities of shrimps, for example, would otherwise be sent to landfill rather than returned to sea. At times when it is mostly weed and rubbish that are removed at the high-pressure wash stage, the material could be collected for disposal.

The geometry of band screens makes them more adaptable than drum screens to fish handling and most FRR development has been based on the band screen format. As it is likely that drum screens will be used in new nuclear stations, the difficulties of drum screen geometry will require attention. The main issue is that the tipping radius of the fish bucket is much larger than on band screens, so fish tend to fall out of the bucket towards the collection hopper over a relatively large arc. Thus, a large proportion of fish can fall back into the screenwell and may have to be handled multiple times before successful removal. Figure 6-23 illustrates this problem with a photograph from one nuclear plant where scaffold boards had been placed in front of the hopper and were seen to collect quantities of fish that missed the hopper.

Figure 6-23 Fish that have overshot the hopper collecting on a scaffold boards inside a drum screen chamber

Where the use of biocides upstream of fish return facilities is critical to security of plant operation, a suitable ecotoxicological risk assessment should be carried out to ensure that any risk of adverse lethal and sublethal effects on fish is minimised, and that the level of ecological impact associated with biocide use is acceptable. Wherever practicable, biocide application should be conducted on inactive legs of the CW system (for example, soak application when no abstraction is taking place) or at times of the year when fish entry is likely to be minimal.

Neither the AFD nor FRR systems used in Britain are capable of protecting small fish of less than around 30 mm in length, which are vulnerable to entrainment.

6.1.7 Special measures at Natura 2000 sites?

No separate regulations or standards on impingement and entrainment apply to Natura 2000 sites but particular attention needs to be paid to species listed in the conservation objectives for the site. Fish species listed under Annex 2 of the Habitats Directive (92/43/EEC) include:

- River lamprey (*Lampetra fluviatilis*)
- Sea lamprey (*Petromyzon marinus*)
- Brook lamprey (*Lampetra planeri*)
- Atlantic salmon (*Salmo salar*)
- Bullhead (*Cottus gobio*)
- Spined loach (*Cobitis taenia*)
- Allis shad (*Alosa alosa*)
- Twaite shad (*Alosa fallax*).

Of these, the three lamprey species as well as loach and bullhead are benthic and likely to be insensitive to AFD techniques but amenable to FRR methods. Conversely, salmon and shad are too delicate for FRR but can be deflected by behavioural methods.

Compliance with the Habitats Directive does not necessarily mean that the project must lie within the designated area of the SAC; if the project might indirectly have an impact on the SAC, such as an abstraction on the migration path of a designated species attempting to reach the SAC, then it will also be subject to the Habitat Regulations[28].

Any development that might affect a Natura 2000 site will be subject to a regulatory assessment process by the competent authority (The Environment Agency of England and Wales). In such an assessment there will be an initial *screening* stage, which seeks to determine whether the proposed development is likely to have a significant impact on conservation features of the European-protected site. Where there is considered to be a significant risk, then the competent authority will undertake an appropriate assessment to evaluate that risk. Where a significant impact is predicted, approval to proceed with the development will be granted only if the proposed project can be modified to mitigate the significant impacts, or is subject to 'overriding public interest' considerations. Further details can be found in various guidance documents (see e.g. http://www.mceu.gov.uk/MCEU_LOCAL/Ref-Docs/EN-HabsRegs-AA.pdf [accessed 08/01/10).

[28] Conservation (Natural Habitats & Conservation) Regulations 1994

6.1.8 Residual impacts

The environmental impact assessment for a new nuclear station should attempt to quantify, among other things:

(a) the proportion of fish present in the approaching CW flow that would be deflected by mitigation measures at the intake structure;

(b) the proportion of impinged fish that would be returned safely to the source water body by FRR measures;

(c) the proportion of entrained early life-stages that would pass back unharmed to the source water body.

Residual impacts will then arise from impinged and entrained fish that are returned to the wild, dead or injured. The residual impact is reduced further by the reassimilation of this biomass back into the food chain, although this is more difficult to quantify.

Residual impacts from a development may be offset by appropriate forms of compensation. These should be in kind, and at least commensurate with the level of residual impact. The California Energy Commission approach provides a model for this (CEC, 2005), in which the residual biomass loss is converted to habitat production foregone (HPF) (Section 6.1.5) and then new habitat to the equivalent or greater area is created. UK examples of best practice for lost estuarine and coastal production are emerging from parallel considerations in various industrial sectors where coastal habitat projects have been funded (Table 6-7).

Table 6-7 Example of large-scale habitat compensation projects in the UK (Scottish Parliament, 2008)

Existing large scale habit projects in the UK	
South Wales-Newport Wetlands Reserve	• 439ha site as compensation for habitat lost to Cardiff Bay Barrage • Countryside Council for Wales, Newport City Council & RSPB Partnership • 50,000+ visitors expected p.a.
Humberside–Alkburgh Flats	• 370ha of intertidal and other habitats created by the Environment Agency • £6.1m project resulting in £23.6m environmental and flood defence benefits
South Yorkshire-Dearne Valley	• restoration of network of sites to create high quality natural habitats • significantly improving area's image and appeal to new residents, businesses and visitors
Teesside-Saltholme	• 380ha former industrial site • will open as RSPB reserve in 2009 with 100,000 visitors p.a. expected • aims to help attract highly skilled, creative individuals to the area • substantial funding from Regional Development Agency - One North East
South East England-Wallasea Island	• RSPB managed project • restoration of 620ha of land back to coastal marshland • adjacent to 115ha DEFRA funded port-development compensation project
Thames Gateway-Rainham	• 353ha RSPB reserve • former MOD firing range • major greenspace for the Thames Gateway

An important consideration in determining compensation is that it should match as closely as possible the loss of species and life-stages. It is normal practice to provide in compensation a larger habitat area, around 1.5 times the estimated HPF figure, to allow for error in the estimate, any underperformance of the replacement habitat and habitat maturation time.

6.2 Effects of cooling towers

6.2.1 Aerial emissions

Pathogens

The primary concern is with *Legionella pneumophila* that was first recognised in 1977 as the causative agent of a form of infective pneumonia. There are at least 14 serotypes and several subtypes that differ in their distribution and pathogenicity. Various estimates suggest that Legionnaires' disease, LD, constitutes about 0.1 to 0.2 per cent of all pneumonias in the UK, but, as pre-1977, it is probable that an unknown number of infections will go unrecognised. The organisms are ubiquitous in natural waters and may have a symbiotic relationship with protozoa in biofilms. For a water system to give rise to risk, the following factors are needed:

- a source of the organisms – invariably present in river make-up water;
- favourable conditions for growth:
  - temperature 20-45˚C;
  - existing microbiofouling – typically a mature bacterial slime;
  - iron or its salts;
  - stagnation;
- a means of generating a respirable aerosol;
- sufficient concentration of infective organisms to provide an effective dose;
- an opportunity to inhale a contaminated aerosol;
- a susceptible individual, typically male and unhealthy, aged 40-70.

To date no link has been established between an infection and power station cooling towers. However there are several cases where a small wet-cooler on an air-conditioning unit has caused an outbreak and so it could be argued that a group of large power station towers could produce an epidemic. Surveys have shown that *Legionella*, at low concentrations, have frequently been present in make-up water and in most areas of the cooling circuit at concentrations of up to10^6 cfu/l[29]. A wide range of concentrations can be found in different parts of the circuit on the same day and there is also day-to-day variation. Initial surveys using samplers suspended from meteorological balloons inside natural draught towers at West Burton (UK) indicated a decrease in viable heterotrophic bacteria, VHBs, with height; no *Legionella* were detected above the eliminators. However, other bacteria from the cooling circuit were found in droplets above the eliminators and in similar concentrations to those in the cooling water. The next question was whether viable bacteria were leaving the tower. This proved difficult to assess but simulations did not suggest that physical stress reduced VHBs so it is unreasonable to assume that *Legionella* would not survive. The ability of *Legionella* to survive in air after leaving the tower is of paramount importance to the assessment of risk. *Legionella* viability in an aerosol improves with increased humidity. In low humidity air osmotic and physical stress, as produced within a contracting droplet, tend to decrease *Legionella's* metabolic activity but, perversely, improves its resistance to further perturbation. The presence of dissolved salts and organic material will tend to decrease the rate of droplet evaporation. The so-called "open air factor" and the effects of UV radiation are critical to the survival of many micro-organisms, but *Legionella* have been found within encysted amoebae where they

[29] cfu – colony forming units (per litre of water). The CEGB studies used plate-counts, following DoH potable water practice and HSC guidelines.

are protected from most stress and this would permit dispersal over long distances. In the 1988 BBC (London) outbreak, infection was acquired 500 m from the source and in Wisconsin infections were reported from two miles away. The BBC outbreak coincided with mild, dull and humid conditions providing optimal humidity, protection from the sun and limiting dispersal of the aerosol by turbulence. To date there have been no cases of legionellosis amongst power station workers related to exposure to drift or to blowout.

Salt dispersal

There are three sources of small droplets in natural draught towers – the water distribution spray-nozzles themselves, secondary droplets formed as the spray hits the top of the pack (film pack) or splashes down through the slats (splash pack) and condensation. The water distribution system is low-pressure and generally produces droplets greater than 200 μm diameter, too large to make any direct contribution to carryover. However with piped rather than open channel, distribution system blocked nozzles will raise the pressure at unblocked nozzles and generate more small droplets. Droplets in the range 50-200 μm diameter are small enough to ascend the tower, but eliminators fitted above the tower's water distribution system and pack should intercept 95 per cent of droplets under 100 μm. Eliminators are similar in design to steam driers and prevent the passage of water droplets by forcing the airflow to make multiple directional changes. However there is a trade-off between removal efficiency and resistance to airflow. A well-designed, well-fitted and maintained eliminator can greatly reduce water loss, solids and biological carryover. Condensation arising from evaporative heat exchange generates droplets under 50 μm diameter (mean over 20 μm) that constitute the visible plume. Theoretical studies, supported by observations inside towers, have revealed the conditions necessary for droplet growth. At all stages droplet size is the most important parameter. The smallest droplets (under 50 μm) remain in suspension but contribute little to the growth of other droplets. Droplets in the range 100-150 μm tend to coalescence with smaller droplets and provide the main source for droplet growth. Air velocities decrease significantly above the "throat" (*vena contracta*) of a hyperbolic tower so many larger droplets (above 250 μm) will hit the tower shell and drip back onto the eliminators as "downrain". Droplets of 240-360 μm in the fastest airflow can be carried out of the top of the tower by a typical 1.0-1.4 ms^{-1} updraft. Direct measurements of carryover using Andersen cyclone samplers and isokinetic samplers inside and outside natural draught cooling towers were attempted at Drax, Yorkshire (for pathogens) and at Ince, Cheshire (for salt).

Growth by coalescence continues in the plume and most detectable precipitation beneath the plume is droplets in the range 200-350 μm. Average drop size decreases with distance from the tower and peak rain-out typically occurs within a few hundred to a thousand metres downwind. If relative humidity is less than saturation considerable evaporation will occur, further restricting rain-out. However the presence of large, and increasing, concentrations of dissolved salts in the droplets inhibits evaporation and can extend the rain-out area. Clouds that may be formed by the plume at some distance from the tower had a greater concentration of aerosol (0.2 μm) droplets (Mertes and Wendisch, 1997).

With induced draught mechanical towers, the air velocity is highest at the exit and these can be more prone to carryover than forced draught towers where air velocities are highest at the air entry. The wet-air path in both of these types of tower is far shorter than in natural draught towers so there is little opportunity for droplet coalescence. However, in view of the consistently high air velocity, it is essential that the eliminators remain intact and in place.

A comprehensive desk and field study of salt dispersal from proposed seawater cooling towers at Winfrith (Dorset) concluded that the projected rate of salt deposition would not be distinguishable from a background of naturally occurring deposition at this exposed, near-coast, site. In all coastal areas, onshore winds carry salt inland and, even when weather systems are not producing onshore winds, a "sea breeze" often develops during the day. The weight of salt carried and the direction and distance will vary from day to day, so any estimate of salt drift from towers has to be detectable against this wide spatial and temporal variability. As with coastal erosion, a single storm event can produce more salt damage to vegetation than decades of "normal" deposition. If salt drift is uniformly spread around a tower, the annual rate of deposition can be up to three orders-of-magnitude less than the rate of natural deposition 600 m inland from the ocean (Parker, 1979). Ince measurements that were part of the Winfrith study found that local deposition was greatest during rainfall.

The main problem usually identified with salt drift is build-up in the soil and scorching of vegetation, with economic effects on forestry and agriculture, although at Winfrith the requirement was to protect a sensitive heathland habitat. Although salt drift was raised as an issue at the planning stage for Deeside, Connah's Quay and for both of the Killingholme CCGT's there has been no evidence of adverse off-site effects attributable to salt drift and no subsequent mention of the topic in the fifteen or so years for which some of these plants have been operating. Earlier, Roffman and Roffman (1973) had used deposition models to calculate possible incremental increases in the salinity of soil, irrigation water and natural fresh waters from natural and mechanical draught salt water cooling towers. The incremental effects of salt deposition from 1,000 MWe towers (circulation rate of 31.5 m^3s^{-1} and 0.002 per cent drift) upon the surrounding soil and water generally were minimal. Some extreme cases may develop under severe weather conditions, but these will be infrequent and will represent a small fraction of the total operating time. Studies around five seawater cooling towers near Galveston (Wiedenfeld, 1978) found levels as high as 1,200 kg ha^{-1} per year within 100 m, decreasing logarithmically with distance to under 300 kg ha^{-1} per year at 434 m. Only 16 per cent was attributable to the cooling towers; the balance was natural sea spray that averages about 250 kg ha^{-1} per year in the study area. There were only slight observable effects in the soils closest to the towers that may eventually lead to salinization and solonization (clay deflocculation due to high sodium levels).

Fog and ice

Plumes of water vapour (fog) can often be seen rising from cooling towers. This occurs when the warm saturated air leaving the tower mixes with ambient air and is cooled to its saturation value. The visibility of the plume depends upon the concentration of micro-droplets that have condensed from the vapour and upon the direction of illumination. Back-lit plumes are dark, and are frequently mistaken by the media as symbols of industrial pollution. Front-lit plumes are bright white. Plumes are bigger and more persistent when the ambient air is at or near saturation. The visible plume may rise until its buoyancy is in equilibrium with the ambient air and it becomes "cloud"; alternatively the visible plume may vanish a short distance above the tower but reappear as cloud at altitude some distance downwind.

In cold weather, droplets in the plume may become supercooled and coat adjacent surfaces in ice. Alternatively, the ground may be below freezing point and ice will form where it contacts the cooling, sinking plume. Icing is mainly a problem associated with low, mechanical draught towers. TRC (2007) indicates the limitations of a modelling approach. Modelling could predict whether the plume water vapour content was greater than or less than the ambient saturation deficit and therefore the potential presence of a condensed plume, but could not estimate plume density, that is, whether or not the plume was barely saturated and would be evaporating. Similarly it would predict ground (rime) icing whenever it predicted a ground fog coinciding with sub-zero (centigrade) air temperatures. These conditions may be conducive to rime ice formation, not actual formation of rime ice, which is a complex process and strongly affected by factors such as density of supercooled liquid water in the plume, plume orientation and wind speed. The model also considered plume abatement (plume mitigation) by diverting 10 per cent of the incoming water through the dry section of a hybrid tower. Heat rejection through the dry section was assumed to be five per cent of total tower heat duty, reflecting the fact that about half of the heat rejection duty of the tower during cool weather is achieved by sensible heat transfer, rather than by evaporation. This abatement is anticipated to result in negligible plume fogging or icing beyond the site.

6.2.2 Visual and aesthetics

Public concern over cooling towers mainly stems from the visual impact of these large structures, especially the commoner hyperbolic natural draught towers, along with their visible vapour plumes. According to Winter (1997), planning authorities have been increasingly reluctant to accept visible plumes since the privatisation of the UK power industry and this would be a key consideration of nuclear plant developers in selecting a viable cooling system.

In order of visual impact, natural draught cooling towers are the most offensive, typically being 100 m tall or higher and invariably with a visible plume, followed by mechanical draught and hybrid mechanical draught and dry cooling towers (all around 40 m in height). Of the last three types, vapour plumes are greatest with the first and absent with dry cooling; hybrid towers reduce the visible plume by heating moist air on heat exchangers before it leaves the tower.

Guidance on assessing the visual impact of aerial plumes is given in Section 3.8 of the IPPC Horizontal Guidance Note (Environment Agency, 2003). Criteria for assessing a visual impact are:

- percentage of daylight time that the visible plume or its shadow extends beyond the site boundary of the plant;
- frequency of impact on surrounding area;
- presence of sensitive receptors.

An impact is only deemed to be insignificant if the visible plume extends beyond the boundary for less than five per cent of daylight hours, the impact is deemed to be small and there are no sensitive local receptors.

6.2.3 Chlorination by-products

The addition of chlorine to water is an instantaneous reaction resulting in an equilibrium mixture of hypochlorous acid (HOCl) and hypochlorite ions (OCl^-):

$$Cl_2 + H_2O \text{ <> } HCl + HOCl$$

$$HOCl \text{ <> } H^+ + OCl^-$$

With sodium hypochlorite:

$$NaOCl \text{ <> } Na^+ + OCl^-$$

$$H^{+} + OCL^{-} <> HOCl$$

there is a 50:50 mixture of the “free chlorine” oxidants HOCl and OCl- at pH 7.5 but the equilibrium shifts towards OCl^{-} with rising pH. Tower circuits tend to stabilise around pH 8.5 which is unfortunate since HOCL is a more powerful oxidant, and hence biocide, than OCl^{-}. On the other hand HOCL is volatile whereas OCl^{-} is not, so at this pH more of the free chlorine will remain in solution. Reactions begin immediately with any ammonia and organic amines that may be in the water, forming inorganic and organic chloramines (“combined chlorine”). Ferrous and manganous ions, sulphides, sulphites and nitrites are simply oxidised whereas organic material may undergo oxidation, addition or substitution reactions. Most of this organic material - such as humic and fulvic acids - is of natural origin and ubiquitous in natural waters. When chlorinated these yield a range of compounds (“chlorination byproducts” or CBPs), some of the most common being haloforms such as chloroform.

Holzwarth *et al.* (1984a & b) modelled the fate of chlorine in intermittently chlorinated wet-tower cooling circuits and identified three pathways for loss:

(a) flash-off or stripping as the air stream through the cooling tower strips HOCl and volatile chlorination products from the water;

(b) purge or blowdown that dumps a proportion of the free and combined chlorine and CBPs into the receiving waterway;

(c) chlorine demand that steadily converts the biocide to inert (non-oxidant) CBPs.

Although the authors made no allowance for any “chlorine” lost in drift (blow out) from the base of a tower, there was good agreement between observed and predicted free chlorine concentrations throughout the chlorination cycle in the tower used for model validation. About 10-15 per cent of the HOCl was stripped on each passage through the tower and overall about 10 per cent of the added chlorine flashed off, two per cent was purged and 88 per cent was “consumed” as demand. Adding acid to reduce the pH from 8.5 to 6.0 increased HOCl flash-off ten-fold, so any benefit to biocidal activity would have been cancelled by greater oxidant loss and by the cost of acid required. Mono-, di- and trichloramines flash-off much faster than HOCl, so their value as biocides in tower circuits is probably negligible. Between 50 (Draley,1973) and 100 (Holzwarth *et al.* 1984a & b) per cent of the dichloramine was volatilized but the chlorine lost in the form of chloroform and other trihalomethanes (THMs) was minor since only some 0.3 per cent of the added chlorine re-appeared as volatile THMs (Pizzie, 1984; Uhler and Means,1985).

A study by Jolley *et al.* (1977) found that a background chloroform level of one $\mu g\ l^{-1}$ in a tower pond, that rose to 38 $\mu g\ l^{-1}$ during chlorination (to a nominal level of two $mg\ l^{-1}$ TRO) had fallen back to 6.2 $\mu g\ l^{-1}$ two hours later. Most of the chloroform was rapidly lost to the atmosphere and it was assumed that other THMs were volatilized, bromo-dichloromethane (BDCM) being more volatile than dibromochloromethane or bromoform. Similar studies (summarised in EPRI, 1986) found that during routine intermittent chlorination the maximum concentration of volatile THMs in the circulating water was 15 $\mu g\ l^{-1}$, with 35 $\mu g\ m^{-3}$ in the vapour plume. During the six to eight hours after chlorination had finished, the concentrations of free and combined chlorine and of THMs in the circulating water fell back to initial values (around two $\mu g\ l^{-1}$).

To put the above into an environmental and health context, these haloform loadings and concentrations should be compared with standards in current legislation. However, the environmental legislation is wholly aquatic and the only human occupational exposure standards (OESs) are for air. Moreover, the WHO air quality standards appear to cover di-halomethanes, not tri-halomethanes such as chloroform.

The UK OES for chloroform is 50 mg m^{-3} for eight-hour respiratory exposure and 225 mg m^{-3} for 10-minute exposure. The US threshold limit value (TLV) for healthy adults, based on eight-hour daily, 40 hours a week of exposure, is 5 to 50 mg m^{-3} (EPRI, 1986). In this 1986 EPRI cooling tower study, chloroform concentrations in the tower plume (35 µg m^{-3}) were two to three orders of magnitude lower than the US TLV whilst at the predicted touchdown point for the plume, chloroform concentrations were calculated to be four to five orders of magnitude lower than TLV (0.003 to 0.05 µg m^{-3}). The total THM concentration at plume touchdown was only 0.05 to 0.8 µg m^{-3}. The spread in values results from the use of different dispersion coefficients and meteorological conditions.

No UK, EC or World Health Organization environmental standard was found for chloroform or other THMs in the air. In these circumstances, a working limit of one-fortieth of the occupational exposure standard is sometimes assumed. It is apparent from the above figures that the predicted total THMs concentration at plume touchdown would still retain a 1,000-10,000 fold safety margin over working limits derived from UK occupational exposure standards (0.8 µg m^{-3} at touchdown compared with an allowable 1.25 or 5.6 mg m^{-3}).

The key European aquatic environmental legislation is the framework Directive 76/464/EEC (Discharge of dangerous substances to the aquatic environment) plus Directive 88/347/EEC that addresses chloroform. The environmental quality standard for chloroform in fresh, estuarine and marine waters (all uses) is 12 µg l^{-1} as a total annual average. However, the applicability of chloroform legislation to bromoform (essentially saline waters) needs to be evaluated (see Coughlan and Davis, 1984). Peak levels measured in the circulating water during intermittent chlorination of freshwater tower circuits may exceed 12 µg l^{-1}, but retention and dilution within the circuit reduces concentrations in the discharge (purge) to within this standard.

Although 88/347/EC is directed at discharges from plants that <u>manufacture</u> chloromethanes, it is worth examining the values set since <u>any</u> plants that discharge chloroform (including "*plants in which cooling waters or other effluents are chlorinated*") will have limits determined by the EC "*at a later stage*". THM loadings calculated by Pizzie for a 500 MWe seawater-cooled unit using low-level chlorination are equivalent to a chloroform discharge of 250 kg per month. This is similar to the permissible monthly discharge from a 40,000 tonne capacity methane chlorination plant. To comply with Article 3(6) of Directive 86/280/EC, member states must ensure that a process involving agitation in the open air of effluents containing chloroform (this presumably could be taken to include cooling towers) should not result in "an increase in pollution of other media, notably soil and air". It is probable that the Commission envisaged cases of intentional volatilisation in order to reduce aquatic discharges. Nevertheless in the context of an environmental statement, the volatility of chloroform <u>should</u> <u>not</u> be emphasised as a plus point when considering aquatic discharges.

6.3 Effects of the thermal discharge

6.3.1 Thermal plume and long-term heat field

The thermal discharge from a nuclear power station is by far the largest licensed discharge from the plant and is designed to be isolated from any liquid discharges arising from the nuclear island. In some cases it may be used to carry other non-hazardous liquid waste, such as site drainage from non-critical areas, water treatment plant waste-water sewage treatment plant effluent and 'grey' water, in order to take advantage of the large dilution factor (see Section 1.1). Environmental effects of the thermal discharge primarily relate to temperature rise and CW system biocide residues and decay products that may be present.

The following definitions will be used in this section:

Thermal discharge: heated water from the power plant cooling system at the point of release into the estuary.

Thermal plume: the short-term heat field in proximity to the thermal discharge where behaviour of the effluent is still influenced by its momentum and/or buoyancy.

Long-term heat field: area of the receiving water body where temperature is influenced by the heated effluent.

Delta-T (ΔT): temperature rise associated with the heated cooling water.

The thermal plume is often visible from above, owing to the different refractive index of the warmer water, and sometimes from differences in turbidity between source and receiving waters. Localised foaming, caused by the breakdown of plankton and seaweed fragments in the cooling water, may also be evident (Figure 6-24). This has sometimes been a cause of concern to members of the public who mistake it for some kind of 'nuclear' release.

Figure 6-24 A thermal plume, made visible by alginate-induced foaming

Important features of the thermal plume are its buoyancy and, in tidal waters, its constant changing of shape and position with the tides. When tidal flows are strong, around the mid-flood and mid-ebb periods, the plume will be elongated and run parallel to the shoreline; at slack high and low water conditions, the plume will expand radially and develop greater width. In an estuarine channel this will cause it to occupy a greater proportion of the channel cross-section. The seabed immediately beneath the plume therefore receives little warming effect.

At an offshore discharge (the majority of UK nuclear stations: Figure 6-23), plume buoyancy, caused by the lower density of the warmer water, causes the heated effluent to rise in an inverted cone towards the surface. The effect of this is to limit the likelihood of contact of the undiluted effluent with the bed. As the plume spreads, the temperature falls rapidly as a result of dilution and loss to the atmosphere. Therefore, if at some point further downstream the plume does make contact with the bed, it will be at a much reduced ΔT.

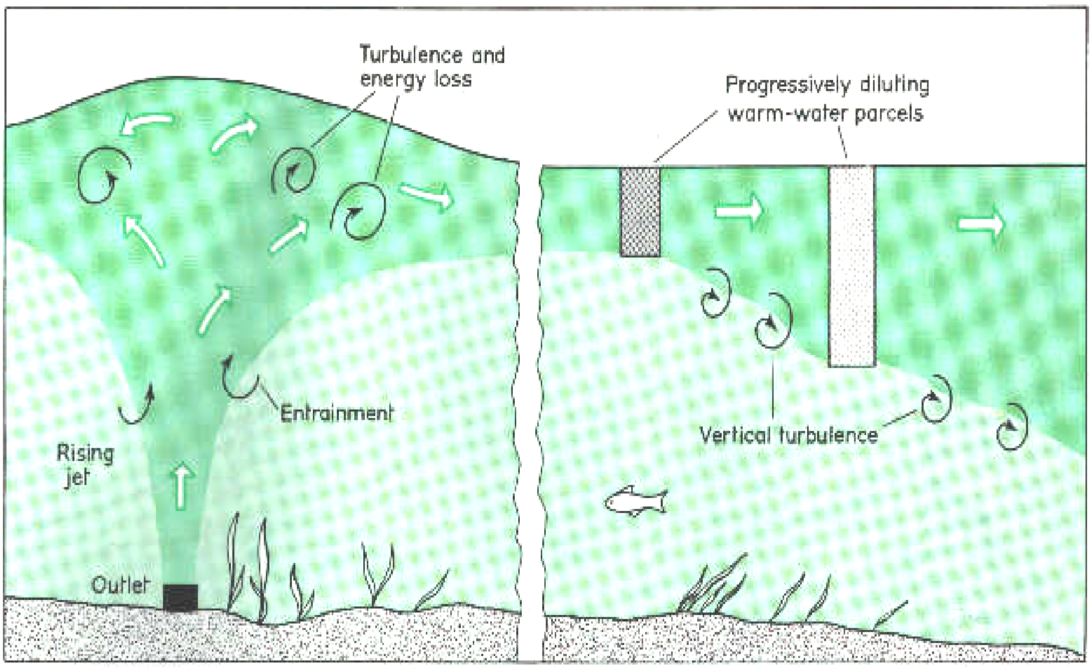


Figure 6-25 Schematic vertical section through an offshore thermal plume showing the hydraulic processes associated with dispersal. Close to the plume, where ΔT is highest, the water rises above the bed, so effects on benthos are usually small here. Further away, the plume mixes with the receiving water and rapidly cools.

Where the CW is discharged across the foreshore (onshore discharge), bed contact of the effluent will be more common and will be greatest at lower tidal levels. The plume will nonetheless become buoyant in these instances when it hits the receiving water body. Examples of direct-cooled stations with onshore discharges include Heysham I & II (nuclear), Marchwood (CCGT) and Kingsnorth (coal). Kingsnorth is an unusual example in that the thermal discharge enters a tidal creek before joining the main Medway Estuary. Such a situation maximises the temperature fluctuations to which any resident biota are exposed.

Other notable features of onshore discharges that cross the foreshore are the potential for scour, especially across mudflats, and for displacing any longshore-migrating fish or other biota into deeper water (sometimes known as the “travelator effect”). This is most likely to affect juvenile fish such as glass eels or flounder, which ascend estuaries along the margins using selective tidal stream transport (Colclough *et al.* 2002).

6.3.2 Water temperature standards

The legal status of water quality standards for temperature applicable to the licensing of new nuclear stations is somewhat unsatisfactory at present. In the past, in the absence of national or European standards for ecosystem protection, temperature guidelines based on the European Freshwater Fish Directive have sometimes been used to provide ad hoc operational standards for estuaries and coastal water, but without legal standing. Also, the EC Shellfish Directive places a 2ºC limit for a rise in temperature within EC-designated shellfisheries and this has sometimes been adopted as a standard for marine waters generally, although it is a guideline value only and is not enacted into English and Welsh law; the directive sets no imperative value for temperature. The UKTAG report on Surface Water Standards and Conditions (UKTAG, 2008) acknowledges, based on consultation responses, that: "*the power industry has pointed to the natural establishment and continued development of clam and oyster beds in transitional waters around the outfalls from power stations although temperature increases greater than 2ºC are routinely observed*", which casts doubt on the validity of this guideline. Further guidance was given under a Review of Consents for the Habitats Directive in which an interim 2°C uplift standard was adopted for marine special protection areas and marine special areas of conservation designated under the Habitats Directive

Efforts have been made to make sense of UK temperature standards to meet requirements of the WFD. Following Department for Environment, Food and Rural Affairs (Defra) guidance, temperature rises caused by power stations will be assessed against draft WFD standards published by UKTAG[30] (2008) on the requirements for coastal and transitional waters to have good ecological status. Table 6-8 lists maximum temperatures that should not be exceeded for more than two per cent of the time (annual 98th percentiles) for each level of ecological status at the edge of a mixing zone. The two per cent allowance takes account of the fact that the plume will spread for a short time around slack-water periods. Transitional and coastal waters will generally support runs or populations of cold-water species such as salmon, sea trout and smelt and therefore the 'cold water' values apply, and will be subject to a 98th percentile temperature limit of 23°C.

An additional requirement of the draft standards is that, outside the mixing zone, a maximum temperature uplift relative to background (ΔT) of +3°C is allowable, except for waters of high ecological status where a 2°C uplift limit is proposed.

Table 6-8 Draft WFD standards against requirements for transitional waters to have good ecological status

	Temperature (°C) (annual 98th percentiles)			
	High	**Good**	**Moderate**	**Poor**
Cold water	20	23	28	30
Warm water	25	28	30	32

[30] United Kingdom Technical Advisory Group for the Water Framework Directive

6.3.3 Chemical pollutants and other interactions with receiving waters

Biocide residues

Terminology

Chlorine in water is variously described as "free", "active", "available", "combined" or "residual" - or by some combination of these adjectives.

(a) **Free (available) chlorine or FAC** is that present as an equilibrium mixture of hypochlorous acid HOCl and hypochlorite ions OCl^-. Both are oxidants. Fugitive elemental chlorine can be ignored.

(b) **Combined (available) chlorine** is oxidants available in (mainly) inorganic chloramines and in other compounds having an N-C link.

(c) **Total (available) chlorine or TAC** is essentially the sum of (a) and (b).

(d) **Residual** is analogous to **available**, but serves to emphasise the concept of a pool of oxy-disinfectant capacity remaining after the initial **demand** (below) has been met. A **residual** may or may not be **stable**.

(e) **Chlorine demand** is defined as the difference between the amount of chlorine added (dosage) and the useful (residual) chlorine remaining at the end of some specified contact period. It is best envisaged as a once-off reaction with a finite quantity of oxidisable substrate.

(f) **Chlorine decay** is a continuing series of reactions that will in time lead to the complete disappearance of all measurable chlorine. Decay is not substrate-limited.

For a chlorine species to be capable of performing as a disinfectant, it must have an ORP (oxy-reduction or redox potential) high enough to oxidise iodide to iodine at pH 7. Fortunately, the most readily available methods for "chlorine" determination do actually measure oxidant capacity and most do so via the stoichiometric iodide/iodine route. For this reason it is preferable to refer to total, free and combined residual **oxidant** rather than to total, free and combined residual **chlorine. Total residual oxidant or TRO** is numerically equivalent to **TAC** as defined above.

When seawater is chlorinated there is another reason for preferring the use of “oxidant”. Seawater contains up to 68 mg l^{-1} bromide from which bromine is displaced when chlorine (or hypochlorite) is added. This yields hypobromous acid (HOBr) and subsequent events are essentially bromine chemistry. The generic term “halogen” is commonly used in place of chlorine and bromine.

For maximum clarity the term **chlorine produced oxidants or CPO** is preferable to **total residual oxidants or TRO**, although given the analytical problems it is doubtful whether the two could be distinguished. Most methods for determining CPO or TRO also detect other oxidants, although not dissolved oxygen, naturally present in the water. These have no antifouling value and for practical purposes are insignificant against the concentration of CPOs (but see below).

Since chlorine reacts with virtually any oxidizable material in water, it follows that its biocidal potential will be short-lived in waters of high chlorine demand. In poor quality waters this can come from reducing agents such as sulphites and sulphides, as well as organic (carbon) material. Organic demand appears to be unrelated to the concentration of chlorine added, suggesting that only some compounds are involved. The low molecular weight carbon fraction increases after chlorination, suggesting scission of macromolecules. Removing organic matter and/or oxidation of metal ions by UV irradiation removes demand. Chlorine is also consumed by addition and substitution reactions with organic matter, particularly humic acids, to form a range of organohalogenated compounds. Exactly what is formed depends upon what is present in the water and this may include anthropogenic inputs (pollutants) such as phenols. These compounds are referred to as chlorine (produced) byproducts **CBPs**. For further details see Khalanski *et al.* (1998).

TRO measurement in relation to standards

A chlorinated discharge of saline water will contain a mixture of chlorine and bromine species and so any standard expressed in terms of chlorine - especially as free chlorine - is unlikely to afford adequate protection to organisms and could be legally unenforceable (Coughlan and Davis, 1984). By the same token any marine toxicological data expressed, or purportedly measured, as free chlorine need substantial qualification, if not rejection.

The standard recently derived for UK marine waters - 10 μg l^{-1} - was therefore expressed as TRO, the sum of 'free' and 'combined' oxidant. This was a maximum allowable concentration MAC. No annual average EAL was proposed because of the lack of chronic exposure data for chloramines and bromamines (Lewis *et al.* 1994). However, this value is at the limit of detection of most practical methods of in-plant and field measurement. Coughlan and Davis (ibid) suggested that seawater chlorination is one of the areas where it is necessary "*to separate the water quality standard from the practical realities and constraints of a means to monitor compliance*" and these exact words appeared in a statement issued following a recent Californian workshop (SWRCB, 2003). In summing up the final (sixth) Chlorination Conference, Roberts (1990) noted "*the near equality of residual chlorine concentrations estimated to be acutely toxic, and those that are measurable. The detection limit for chlorine residuals has remained essentially unchanged over the past decade.*" He was in fact referring to laboratory analytical methods; the detection limit of field methods and, especially, continuous monitoring equipment has been virtually unchanged since 1950. There is an additional constraint. For more than 30 years, workers have been aware of natural "background" oxidants that can be found at concentrations of 30 μg l^{-1} (Eppley *et al*,1976) or up to 60 μg l^{-1}, even in mid-oceanic water (Carpenter and Smith, 1978). These levels greatly exceed the interim UK EQS (10 μg l^{-1}), let alone the proposed/potential 1.0 μg l^{-1}.

Speciation and fate of residual oxidants

As noted above, the chemistry of chlorine in saltwater is complicated by the presence of bromide. In the absence of ammonia, chlorine will react rapidly with bromide to form hypobromous acid (HOBr) that dissociates to form hypobromite ions (OBr-) and hydrogen ions (H+), a reaction which is influenced by pH. A large proportion of hypobromous acid remains undissociated at seawater pH. These free halogens may have only a fleeting existence when ammonia is present, with which they combine to yield halomines - predominantly dibromamine ($NHBr_2$). Dibromamine is more persistent than monobromamine, although the degradation rate depends on pH, temperature and ammonia concentration. Together these species constitute the free and combined oxidants responsible for biofouling control. They are non-specific and will react with virtually any oxidizable material – the so-called chlorine demand - in the water. Typically some 48 per cent of the initial oxidant is lost within one minute to demand reactions. From this point oxidant species undergo decay reactions. These are not substrate limited and are thought to be auto-catalytic or catalysed by metals, and they eventually remove all measurable oxidant.

Any quality standard or legislation referring to "chlorine" or, particularly, "free chlorine" has little relevance to most marine situations. Furthermore, the most widely-used measurement methods add a trace quantity of iodide to the sample, after determination of the free species, to enable the combined species to be estimated. Seawater already contains iodide so there is breakthrough of combined species into the "free" determination – another reason for restricting measurement and standards to TRO.

Apart from providing oxidants, chlorine can halogenate organic molecules to form so-called chlorine byproducts or CBPs. These reactions contribute to the chlorine demand of the water but the resulting CBPs have little use in biofouling control and some may have chronic toxicological significance in the environment.

Chlorine byproducts (CBPs) and bioaccumulation potential

Addition and substitution reactions with natural organic material such as humic and fulvic acids and with some anthropogenically-derived material can yield a variety of volatile and non-volatile chlorinated compounds or CBPs. These include trihalomethanes (THMs) such as chloroform (in brackish water) and bromoform; halogenated phenols, such as tribromophenol and haloacids, such as chloroacetic acid. Although only a few percent of the applied chlorine is involved, these reactions are highly significant because of the persistence and potential mutagenicity of some of the products. In general, the higher the organic content of the source water, the higher the potential for byproduct formation. Whether this potential is realized will depend primarily on the applied chlorine dose, as well as on the extent of competing reactions that lead to the consumption of chlorine. Seawater contains only around 0.5 mg l^{-1} total organic carbon (TOC), about one-tenth of that found in typical river water. About 90 per cent is in the form of humic and fulvic acids. Farm run-off and sewage effluent greatly increases the concentration and variety of substrates and the latter may also introduce trade effluents such as phenols and other aromatics that may give rise to polychlorinated biphenyls and chlorophenols. CBPs are more likely to accumulate in sediment, organic detritus and biota than products such as inorganic chloroamines. Bioaccumulation of the oxidant species is improbable owing to their reactivity and rapid degradation and the low molecular weight CBPs, such as trihalomethanes, will not accumulate appreciably on sediments or in biota.

The literature details hundreds of CBPs but much of this comes from the USA where it is common practice to chlorinate wastewater (sewage effluent). However the reaction conditions - high concentrations of organic material and high concentrations of chlorine - are markedly different from those found in low-level chlorination of seawater.

Effects of chlorinated (halogenated) byproducts (CBPs)

There are two aspects to be considered here: direct toxicity and bioaccumulation.

The most abundant CBP in saline waters is bromoform. This is less acutely toxic and less susceptible to bioaccumulation than chloroform, formed mainly by chlorination of fresh or brackish water. For example a recent comprehensive evaluation of acute and chronic chloroform toxicological data, using 10 datasets for algae, 17 for aquatic invertebrates and 23 for fish, indicated a predicted no-effect concentration (PNEC) of 72 µg l^{-1} that included a fifty-fold safety factor (Zok *et al.*1998). Environmental levels invariably are much lower than this.

By contrast, bromoform concentrations above 32 mg $l^{-1}$are needed to cause a 50 per cent reduction (EC_{50}) in the cell division of four species of marine phytoplankton, *G. halli*, *I. galbana*, *S. costatum* and *T. pseudonana* (Erikson and Freeman,1978). Larval oysters *Crassostrea virginica* (a standard ecotox assay subject) appear to be particularly sensitive to bromoform, with lethal and sub-lethal effects reported at concentrations of 0.05 mg l^{-1} (50 µg l^{-1}) and below, and with a 48-hour LC_{50} (lethal concentration) of one mg l^{-1} (Stewart *et al.* 1979). Gibson *et al.* (1979a) *Protothaca staminea* (littleneck clam) closed up their shells and retracted their siphons, thereby avoiding exposure at concentrations of 300-400 mg l^{-1}. At concentrations of 800 mg l^{-1} they died. *C. virginica* (American oysters) and *Mercenaria mercenaria* (hard-shell clams) ceased filtering and closed their shells at under 10 mg l^{-1}. At 27 mg l^{-1} there were no mortalities during exposure but some died shortly afterwards. The LC_{50} was estimated to lie between 40 and 150 mg l^{-1} for both species. Shrimp (*Farfantepenaeus aztecus*) was more sensitive, having a 96-hour LC_{50} of 26 mg $l^{-1}$and a sub-lethal response was apparent. Within 60 seconds the shrimps had moved as far away from the bromoform source as possible and later lay on their sides (Gibson *et al.*1979a). These authors found considerable difficulty in maintaining experimental concentrations due to volatility. Ali and Riley (1986) found an increased respiration rate in adult *C. virginica* exposed to 25 µg l^{-1} but their feeding rate was reduced. Bromoform uptake was rapid but on return to clean water depuration was complete within 96 hours although there appeared to be irreversible damage (decrease in size) to the gonads. A fish (menhaden - *Brevoortia tyrannus*) had an LC_{50} of 12 mg l^{-1} (Gibson *et al.*1979a).

The most abundant haloacetic acid (HAA) is monochloracetic acid. This is completely ionised at the pH of most natural waters, is not very volatile but is biodegradable: 73 per cent is converted to CO_2 in eight to 10 days at 29º C (Boetling and Alexander, 1979). Its freshwater toxicity is comparable to that of chloroform, with effects at concentrations above one mg l^{-1}. No data were found for saline waters. Trichloracetic acid (TCA) is produced during pulp and paper manufacture and by chlorination of potable and cooling water. An estimated 55,000 tonnes annually enters the aqueous environment and, being highly soluble and fully dissociated in water, it should be expected to remain in that phase. Atmospheric TCA is washed out by rain and snow. In Tokyo Bay the seawater had a mean concentration of 1.7 µg l^{-1}, significantly higher than in European lakes and rivers. However, the source appeared to be polluted rivers and drainage from the Tokyo metropolitan area (Euro Chlor, 2002a).

Trichlorophenol (TCP) is slightly volatile and photo- and biodegradable. In urban wastewater biodegradation is complete within seven days (Tabak,1981). Toxicity is moderate, with a 24-hour LC_{50} of 10.0 mg l^{-1} for goldfish *Carassius auratus* and a 96-hour LC_{50} of 0.1 to 1.0 mg l^{-1} for the fathead minnow *Pimaphelas promeles*. No data were found for saline waters.

Although concentrations of CBPs in most coastal areas may be low, the potential for bioconcentration and bioaccumulation must be considered. This occurs when organisms are unable to excrete a compound and it accumulates in tissues. From the data available, few of the potential CBPs are chemically suited to bioaccumulation. Bromoform, the most abundant CBP in saline waters, is less suited than is chloroform. Apart from straightforward exposure, further biomagnification occurs during subsequent grazing and predation by organisms higher up the food chain, leading potentially to man. Strictly, bioconcentration refers to uptake from water, typically via the gills or body surface whereas bioaccumulation includes dietary sources. The bioconcentration factor BCF is calculated from the concentration measured in tissues compared with the concentration in the water to which the organism has been exposed; this latter is often uncertain since it may fluctuate widely. Chemicals are usually considered to be bioaccumulative when experimentally derived BCF values exceed 2000. The tendency of an organic chemical to accumulate and concentrate in organisms often depends on its hydrophobicity or lipophilicity. This can be predicted from the logarithm of the ratio of a chemical's equilibrium solubility in *n*-octanol and in water – the octanol-water partition coefficient, log K_{ow}. For some purposes, as in ecotox screening, log K_{ow} can serve as a practical surrogate for BCF[31].

Compounds having log K_{ow} >4.5 are at high risk of bioaccumulation whilst those <4.0 are at low risk. The bioaccumulation potential of the trihalomethanes appears to be low, compared to many chlor-organic compounds such as pesticides.

Table 6-9 Bioaccumulation potential of some chlorine produced byproducts (based on Chemical Database Management System, Chemwatch Package 2004/1)

Trihalomethanes THMs	**Potential**
Chloroform	log K_{ow} 1.97
Bromoform	log K_{ow} 2.37
Bromodichloromethane	No data
Dibromochloromethane	log K_{ow} 2.24
Haloacetic acids HAAs	
Monochloracetic acid	log K_{ow} 0.22
Dibromoacetic acid	No data
Dichloroacetic acid	log K_{ow} 0.14 -1.39
Monobromoacetic acid	No data
Trichloracetic acid	log K_{ow} <1 to 1.6

The log K_{ow} of the data in Table 6-9 are all well under 4.0, which implies that these CBPs are unlikely to be bioaccumulated and biomagnified along the food chain. Actual bioaccumulation data for chloroform are contradictory and it appears that slight to moderate bioaccumulation may occur in some aquatic organisms. Based on bioaccumulation studies and estimated BCFs, bromoform appears to have less potential for bioaccumulation, despite its higher K_{ow}.

Dissolved oxygen

Dissolved oxygen (DO) levels in waters receiving a thermal discharge can potentially be affected in several ways.

First, the solubility of oxygen in seawater depends primarily on salinity and temperature (Table 6-10) so raising the temperature will lower the oxygen solubility; however, solubility will only affect the actual DO level when the the ambient level is at or close to saturation. DO saturation levels will in any case vary naturally with season and state of the tide. For example, for a water temperature of 16°C and salinity of 34 ppt, the saturation concentration of DO is 8.02 mg l^{-1}, and a 1°C rise in temperature results in a fall in saturation concentration of less than 0.2 mg l^{-1} (USEPA, 1985). Within the thermal plume greater rises in temperature occur and lead to greater changes in saturation concentration. An 8°C rise leads to a reduction in saturation concentration of 1.1 mg l^{-1} with corresponding potential change in DO concentration, depending on the balance of mixing, re-aeration and possible degassing occurring within the plume.

[31] Not be confused with the BCF in the relationship **log BCF = 0.8 log (K_{ow}) – 0.52** derived from K_{ow} and water solubility (Isnard and Lambert, 1988)

Table 6-10 Effect of temperature and salinity on DO solubility (from Turnpenny, Coughlan and Liney, 2006)

Temperature ^{o}C	Solubility of oxygen		Correction factor for salinity: factor to be subtracted
	Freshwater mg l^{-1}	Seawater (35 ppt) mg l^{-1}	
0	14.6	11.3	0.0925
5	12.7	10.0	0.0771
10	11.3	9.0	0.0653
15	10.1	8.1	0.0559
20	9.1	7.4	0.0481
25	8.2	6.7	0.0415
30	7.5	6.1	0.0362

A second effect is caused by biological and chemical oxygen demand (together sometimes known as "effective oxygen demand" or EOD) accelerated at higher temperatures. These processes are responsible for oxygen sag that is often seen during the warmer summer month in organically polluted estuaries[32].

The third effect occurs when oxygen-depleted water becomes re-aerated within power stations after passing over weir or cascades (such as seal weirs) or from the action of natural draught cooling towers. Prior to the clean-up of many British rivers in the 1970s, the re-aeration action of cooling towers was shown to bring major improvements in downstream DO levels where upstream levels were under 4 mg l^{-1}; where upstream levels were higher, small reductions in downstream DO were seen (Langford, 1990).

Depletion of DO in thermal discharge zones of any UK nuclear plants has not been a notable concern in the past, probably since none has been built on an estuary with heavy organic pollution. Where this may be a concern in any new nuclear build, appropriate water quality modelling should be carried out (see Section 6.3.4). Table 6-11 shows the draft WFD standards that will be applied to transitional and coastal waters.

Table 6-11 Draft WFD dissolved oxygen standards for transitional and coastal waters of different ecological status (UKTAG, 2008)

	Freshwater	Marine	Description
	Fifth percentile (mg l^{-1})		
High	7.0	5.7	Protects all life-stages of salmonid fish
Good	5.0-7.0	4.0-5.7	Resident salmonid fish
Moderate	3.0-5.0	2.4-4.0	Protects most life-stages of non-salmonid fish
Poor	2.0-3.0	1.6-2.4	Resident non-salmonid fish, poor survival of salmonid fish
Bad	2.0	1.6	No salmonid fish. Marginal survival of resident species

Ammonia

Ammonia is present in all natural waters, generally at low concentrations that are derived primarily from mineralisation (breakdown) of organic nitrogen, denitrification

[32] See e.g. Bayshore Power Plant study:
http://www.westernlakeerie.org/bayshore_thermal_fish_3163b_etc.pdf

(reduction of nitrate) and excretion by aquatic organisms. However anthropogenic sources such as sewage treatment outfalls and agricultural run-off predominate in most tidal waters. Of all the water quality parameters that affect aquatic life, ammonia is probably the most important after dissolved oxygen. Concentrations are usually measured as total ammonia nitrogen ("ammonia as N") that is an equilibrium mix of free un-ionised ammonia (NH_3) and ammonium ion (NH_4^+):

$$NH_4^+ + H_2O <> NH_3 + H_3O^+$$

The chemical speciation of ammonia is important to its toxicity, since un-ionised ammonia is much more toxic than ammonium ion. Consequently, most ammonia regulations are written in terms of NH_3 although current measurement techniques only measure total ammonia. It is therefore necessary to know the physico-chemical parameters that determine the relative proportions of the two species; these include temperature, pH, dissolved oxygen and salinity. The proportion of un-ionised ammonia, and hence toxicity, increases with pH and temperature but decreases with increasing salinity (US EPA, 1989, 1999; Alabaster *et al.*, 1979; Moreira da Silva, 2009). Of these factors, pH is the most and salinity the least important (USEPA, 1989). In a review of the effects of ammonia on marine and estuarine organisms, Nixon *et al*. (1995) note that acute toxicity to fish increases at low dissolved oxygen concentrations in both fresh and saltwater.

The mechanisms by which pH, temperature and salinity affect ammonia toxicity are not well understood. Although ammonium ion is not as toxic as un-ionised ammonia, it can be important when present in high concentrations: in fresh water it is thought that the pH dependence is due to joint toxicity of un-ionised ammonia and ammonium ion. Temperature affects the proportion of un-ionised ammonia and may also affect ammonia toxicity via its effects on a fish's membrane permeability, endogenous ammonia production and other physiological processes. In freshwater the relationship between the toxicity of un-ionised ammonia and pH and temperature is similar for most species. However, in saltwater there is no evidence that temperature or salinity have a major or consistent influence on the toxicity of un-ionised ammonia. It has also been found that whilst the acute toxicity of ammonia at a given pH depends upon the fish species, chronic toxicity is more dependent on temperature. Milne (2004) proposed that the pH differential across a fish's gills (he used rainbow trout as a model) was crucial for ammonia excretion. A fish migrating from fresh to saltwater would need either to significantly reduce its metabolic production of ammonia or the external ammonia concentration would have to be approximately one-sixth that of the freshwater.

The UK Environment Agency's catchment model SIMCAT (Environment Agency, 2008) describes the "decay" of ammonia in water with time by a first-order equation:

$$\frac{dC}{dt} = -k_T C$$

where C is the ammonia concentration and κ_T the rate constant.
The rate constant is temperature dependent:

$$k_T = k_{20} 1.072^{T-20}$$

Cooling water discharges can increase water temperature and salinity (if the station is fitted with desalination equipment) in the mixing zone and for some distance beyond. The slight increase in salinity might adversely affect some species even though the ammonia toxicity would decrease. Although ammonia toxicity increases with temperature, ammonia concentration would be declining through faster decay.

Alkalinity and pH

The carbonate-bicarbonate equilibria that determine pH and alkalinity in seawater are slightly influenced by temperature, with a 0.0114 pH unit change per degree Celsius (Gieske, 1969). Shifts in the equilibrium position associated with temperature rise will be reversible once the temperature falls as a result of heat loss and mixing. Any effect outside the mixing zone will therefore be localised and negligible with respect to ecological effects.

Salinity

Salinity differences associated with the CW discharge could arise from:

- discharge of cooling water abstracted from a region where salinity differs from that of the receiving water;
- concentration of salts in purge water from seawater cooling towers (indirect cooling only);
- concentration of salts within any desalination facility provided to supply freshwater to the site.

In any of these cases the associated salinity change would be small and unlikely to be detectable beyond the mixing zone.

Heavy metals

Langford (1990, p.98-99) refers to cases of zinc and copper contamination in discharges from US power plants and although copper alloys have been used in the UK, the use of titanium in modern CW condensers eliminates this source. Titanium itself does not dissolve in seawater and presents no toxic risk. Other sources of heavy metals at large power stations relate principally to burning fossil fuels and scrubbing the resulting flue-gases, therefore are not associated with nuclear generation.

Any use of copper elsewhere in the cooling system, for example in screenwell surface treatments or within intake screen materials (Cu/Ni alloy) would potentially give rise to copper within the discharge but dissolution rates are low. Prior to the use of Cu/Ni screens at the new Thames Gateway Desalination Plant (Thames Water Utilities Ltd), an investigation of copper levels in the screened water showed undetectable levels and led to approval of their use by the Drinking Water Inspectorate.

Speciation of contaminants

Changes in water temperature, along with other associated water quality changes (such as DO, pH) are known to affect the speciation and biological availability of contaminants held within bed sediments. Where a thermal discharge enters a heavily urbanised estuary or coastal water body, this aspect may need to be considered. For example, a recent study undertaken in Long Island Sound, New York by Beck and Sañudo-Wilhelmy (2007), showed that copper cycling between sediments and the water column varied seasonally, concentrations of labile copper showing a positive correlation with water temperature and a negative correlation with DO. The authors suggest that warming could increase the biological availability of copper.

Suspended sediments

The settling rate of sediments in water is described by Stoke's Law, which includes terms of water density and viscosity, both of which are influenced by temperature. In theory, therefore, rates of sedimentation could be affected by water temperature in a thermal discharge. In practice, however, the presence of the outfall structure and the potential scouring effects associated with discharge currents are likely to outweigh settlement rate effects. It is significant that Langford's (1990) treatise on thermal discharge effects makes reference in this context only to scour effects.

Eutrophication

Higher water temperatures may accelerate nutrient cycling and growth rates of algae. This may raise concerns over development of eutrophic algal blooms, including those of toxic species. Development of large mats of blue-green algae around the Turkey Point plant outfall in Florida was associated with the thermal discharge but blue-greens favour temperatures in the mid-thirties Celsius (GESAMP, 1990) and are therefore unlikely to develop in UK coastal waters where ΔT values are kept below the WFD uplift standard of 3°C. Entrainment passage of algae within the cooling water will depress viable algal numbers within the plume, which will offset any tendency for greater numbers due to nutrient and warming effects. Langford (1990) found no evidence of toxic algal blooms being enhanced by any thermal discharge. In a review of thermal discharge effects on the marine environment, GESAMP (1984) identified major shifts in algal community composition only in association with ΔT values of 7 to 10°C, well above values that would be allowable in UK waters beyond the mixing zone.

Microbial activity

The activity of waterborne bacteria and fungi generally increases with water temperature, leading for instance to the enhanced biological oxygen demand mentioned above. Like communities of other organisms, microbial community structure shifts according to temperature, reflecting the preferences and tolerances of different species within the mix (Langford, 1990). In addition to effects on DO level, the consequences of such changes may include changes in prevalence of human and fish/shellfish pathogens, notably *Vibrio* spp. and pathogenic amoebae. Human pathogens would be of most concern for discharges close to bathing beaches or areas used for water-contact sports and recreation. Fish/shellfish pathogens such as *Vibrio*, in the context of cooling water, primarily become a problem when the warm water is used for aquaculture. The combination of high stock densities, higher temperature and high organic loadings creates favourable conditions for pathogens to flourish.

In power stations that use chlorine-related products for biofouling control, the presence of oxidant residues in the discharge will act as a disinfectant and reduce pathogen levels, much as occurs when sewage effluents are chlorinated to reduce pathogen risk on bathing beaches.

6.3.4 Thermal discharge modelling

It has been the practice for many years for new power station developments to model the spatial and temporal characteristics of the heat fields, and sometimes other water quality characteristics, within the receiving waters. Modelling provides data for assessing the environmental effects of the discharge and for predicting regulatory compliance with water quality standards. This will undoubtedly be a requirement for new nuclear stations.

This section gives an overview of the computational/mathematical models available to predict the thermal and chemical impacts of cooling water discharges. The exact modelling requirements of thermal and chemical discharges will be site-specific (such as consideration of the spring-neap tidal cycles for discharges into coastal waters) and seasonal effects will be considered. This section discusses the general role of modelling and readily available models, including their suitability for different water environments. The pros and cons of models are outside the scope of this overview.

Role of modelling

Modelling can be used to predict the potential water quality and thermal impacts of cooling water discharges on the water environments under various ambient conditions (such as tides, seasonal river flows, water quality and temperature). It can aid the design of cooling water systems (such as discharge configurations and operating conditions, optimization of cooling water intakes in terms of water temperature and silt content). Modelling also enables the necessary mitigation measures to be adopted.

Modeling predictions can be used to inform BAT (best available techniques) analysis. Modelling allows comparison between different cooling water options. For example, models can be used to simulate the fate of excess chlorine from discharged cooling water (biocide in the form of added chlorine or electro-hypochlorination) and concentrated brine plume as a byproduct of electro-hydrochlorination. Predictions can be used to compare the environmental impact of different antifouling options.

Type of models

Models for predicting the water quality and thermal impacts of discharges into water environments consist of a hydrodynamic module and water quality module. The hydrodynamic module predicts the flows and depth of water across the modelled area. It provides the hydrodynamic information necessary to simulate advection and dispersion of pollutants by the water quality module. Most water quality modules simulate the chemical reactions that take place within the water body. Some also simulate the biological fates of pollutants (such as decay of E. Coli) and pollutants' chemical behaviour between water and sediment phases (such as metal partitioning).

Available methods range from simple dispersion calculations to sophisticated and computationally intensive models. Some models are designed for the initial dilution and mixing zone analysis in the near field of a discharge while others are designed to simulate the advection and dispersion of heat and chemicals beyond the mixing zone.

Models vary in terms of the physical and chemical processes they can simulate and also the dimensionality of models. For example, the configuration of discharges (or the discharge momentum) has a considerable effect on the density currents. Thermal discharges affect the density of currents as well as other water quality parameters (such as reactions rates of water chemistry, primary productivity and nutrient cycling). Saline discharges also affect density currents. Some models account for the aforementioned effect of thermal and saline discharges explicitly while others do not.

Models vary in their dimensionality. There are 1-D (one-dimensional), 2-D (two-dimensional) horizontal, 2-D vertical and 3-D (three-dimensional) models. A 1-D model covers a length scale, normally down the middle section of a river channel or estuary. The choice of a 1-D model assumes that the water is well mixed laterally and vertically.

A 2-D model covers two length scales, which can be along the length and width of a river channel/estuary (a 2-D horizontal model, assuming the water body is well mixed vertically but not laterally), or along the length and depth of a river channel/estuary (a 2-D vertical model, assuming the water body is well mixed laterally but not vertically). A 3-D model covers the length, width and depth of a water body.

The model chosen for a study should be suitable to answer the questions posed in the study and appropriate to the situation it is applied to. Furthermore, a model needs to be calibrated and validated before its predictions can be relied on in decision-making.

Initial dilution and mixing zone models

H1 dilution factors

IPPC H1 guidance (Environment Agency, Environment and Heritage Service and Scottish Environment Protection Agency, 2003; Environment Agency, 2008a; Environment Agency, 2009a) gives simple methods of estimating the impacts of discharges on water environments. Simple mass balance equations using dilution factors are given for discharges of chemical substances to inland rivers, freshwater- and saline-dominated parts of estuaries, and coastal waters (see Table 6-12). H1 also gives details of direct toxicity assessment techniques to assess impacts of highly complex discharges (for example, when the chemical composition is not known).

Table 6-12 Simple mass-balance equations for discharges to various water environments (Environment Agency, Environment and Heritage Service, Scottish Environment Protection Agency, 2003; Environment Agency, 2008a; Environment Agency 2009a)

Water environment	Mass-balance equation
Inland rivers	$PC = \frac{EFR \; x \; RC}{EFR + RFR} \; x \; 1000$
Freshwater-dominated parts of estuaries	$PC = \frac{EFR \; x \; RC}{DRE} \; x \; 1000$
Coastal waters or saline-dominated parts of estuaries	$PC = \frac{EFR^{2/3} x \; RC}{DRC} \; x \; 1000$

where PC = concentration (μg l^{-1}) of chemical substance in rivers from cooling water discharge (process contribution without taking into account background conc.)

EFR = flow rate of cooling water discharge ($m^3 s^{-1}$)
RC = concentration (mg l^{-1}) of chemical substance in the cooling water discharge
RFR = river flow rate ($m^3 s^{-1}$).

DRE = Dispersion rate ($m^3 s^{-1}$) of the fresh-water dominated parts of estuaries
= 2.4 for low nominal dilution conditions
= 5 for medium nominal dilution conditions
= 10 for high nominal dilution conditions

DRC = Dispersion rate ($m^2 s^{-2/3}$) of coastal waters or saline-dominated estuary
= 2.5 for low nominal dilution conditions in coastal waters
= 8 for medium nominal dilution conditions in coastal waters
= 25 for high nominal dilution conditions in coastal waters
= 2.4 for low nominal dilution conditions in saline-dom. parts of estuaries
= 5 for medium nominal dilution conditions in saline-dom. parts of estuaries
= 15 for high nominated dilution conditions in saline-dom. parts of estuaries

The mass balance equation given in Table 6.12 for inland rivers can be used for impact assessments other than the initial dilution analysis. Equations for freshwater-dominated parts of estuaries, coastal waters and saline-dominated parts of estuaries are for the derivation of the 95th percentile initial dilution (reduction in concentration the discharge will receive between point of release and open sea surface for 95 per cent of the time).

ELSID

ELSID has been developed by the Environment Agency to calculate dilutions of discharges into still or tidal waters. It performs Monte-Carlo simulations of initial dilution to calculate the 95th percentile compliance (Scottish Environment Protection Agency, 2006).

VISUAL PLUMES (VP)

Visual Plumes (VP) is a Windows-based mixing zone modelling application developed by the US EPA (United States Environmental Protection Agency). It is designed to replace the DOS-based PLUMES program. Like PLUMES, VP can be used for plumes discharged into fresh and marine waters.

VP supports initial dilution models that simulate surface water jets, single and merging submerged plumes in arbitrarily stratified ambient flows. Predictions include dilution, rise, diameter, and other plume variables. VP has a bacterial decay model and a conservative tidal background pollutant build-up capability. Hence it is useful for mixing zone analyses and water quality applications.

CORMIX

CORMIX (CORnell MIXing Zone Expert System) is a model for the analysis, prediction and design of chemical and thermal discharges into all types of ambient water bodies, including small streams, large rivers, lakes, reservoirs, estuaries and coastal waters. The model focuses on predictions of the geometry and dilution characteristics of the initial mixing resulting from different discharge configurations so that compliance with regulatory constraints can be evaluated. It can be used to simulate near-field thermal effects/temperature distribution of submerged single-port discharges (CORMIX 1), submerged multiport diffuser discharges (COMIX2) and buoyant surface discharges (CORMIX 3). It also predicts plume behaviour at large distances and can be used as a first-order screening and design tool. However, it does not simulate explicitly changing conditions such as tides.

CFX-5

A more recent development has been the ability to use 3-D computational fluid dynamics models such as the ANSYS CFX-5™ model, such models now being capable of handling free-surface flows. Though more costly, they are subject to fewer constraints and offer a flexible approach to evaluating plume behaviour with different outfall configurations.

Beyond the mixing zone

A wide range of models are available for the simulation of thermal and water quality impacts beyond the mixing zone of discharges. They vary in complexity and range of applications. Most models use a hydrostatic pressure assumption in modelling flows. Unless the predictions from a mixing zone model are used as inputs to these models, this hydrostatic assumption can result in inaccurate predictions of the thermal and water quality impacts of buoyant plumes not only in the mixing zone, but also outside the mixing zone and well beyond.

WARNB, MCARLO and AMMONIA

WARNB (Warn-Brew Method) and MCARLO (Monte-Carlo Simulation) are computer programs publicly available from the Environment Agency. They use mass-balance equations and combining distributions to calculate the river quality downstream of a continuous discharge or the discharge quality required to achieve specific water quality targets in rivers (Environment Agency, 2009b). The water quality determinands considered include Biochemical Oxygen Demand (BOD) and Total ammonia. WARNB assumes that the data distributions are log-normal while MCARLO can be applied to other types of data distributions. AMMONIA uses mass balance equations, Monte-Carlo simulation and water chemistry data to calculate the impact of discharges on un-ionised ammonia and total ammonia in rivers (Environment Agency, 2009b).

SIMCAT

SIMCAT is a mathematical model developed by the Environment Agency to calculate the impact of discharges and diffuse inputs into inland rivers on water quality throughout a catchment (Environment Agency, 2008b). It is a one-dimensional, steady state model. It uses Monte-Carlo simulations to mix discharges and diffuse inputs with river waters and then routes flow in the river down through the catchment, applying water quality transformation processes en route.

SIMCAT can be used to model concentrations of chloride, biochemical oxygen demand, total organic carbon, ammonia, dissolved oxygen, phosphate and nitrate. SIMCAT does not simulate river temperature directly but temperature is used to adjust the rate of natural purification, saturation concentrations of dissolved oxygen and decay rate of biochemical oxygen demand, ammonia and so on. It calculates summary statistics of water quality like means and percentiles, allowing the predictions to be compared directly with water quality standards defined as means and percentiles.

ISIS Water Quality

ISIS Water Quality (developed by HR Wallingford and available from Halcrow) is a module add-on to ISIS Professional that can be used to model the advection/diffusion of conservative and decaying pollutants, water temperature, cohesive sediment transport, interaction of quality determinands with sediments, phytoplankton and pH. It can be used for water quality studies in rivers.

QUAL2K

QUAL2K (or Q2K) is a river and stream water quality model developed by the US EPA. It is a modernized version of the model QUAL2E. QUAL2K simulates point and non-point loads and abstractions of heat and mass. It can be used to simulate conventional pollutants (such as nitrogen, phosphorus, dissolved oxygen, BOD), pH, pathogens and periphyton.

MIKE

MIKE11, MIKE21 and MIKE3 are modelling systems developed by DHI (Danish Hydraulic Institute) for hydrodynamic modelling. MIKE11 (a one-dimensional modelling system) can be used for rivers and reservoirs or lakes if it is coupled with a module MIKE11 Stratified. MIKE21 (a two-dimensional modelling system) and MIKE3 (a three-dimensional modelling system) are ideally suited for marine environment (estuaries, coastal waters and sea). MIKE3 can also be used for lakes (stratified waters). The non-hydrostatic pressure formulation of MIKE3 hydrodynamic module allows for an accurate simulation of vertical acceleration of flows. This would improve predictions of the impact of thermal discharges both within and outside the mixing zone.

The basic hydrodynamic models of MIKE coupled with various modules can be used to simulate physical, chemical or biological processes in the water environment. The AD (advection-dispersion) module is typically used in cooling water studies to simulate the transport, dispersion and decay of dissolved or suspended substances. ECO Lab is a module for ecological modelling that can be coupled to MIKE11, MIKE21 and MIKE3. The ECO Lab integrated AD Module also calculates the salinity and water temperature.

WASP

WASP/DYNHYD is a US EPA generalized modelling framework that simulates the transport of heat and fate of contaminant (biochemical oxygen demand, dissolved oxygen dynamics, nutrients, bacterial contamination and toxic chemical movement) in surface waters. The DYNHYD model is a simple hydrodynamic model that simulates variable tidal cycles, wind, and unsteady inflows. It produces an output file that can be linked with WASP (Water Quality Analysis Simulation Program) to supply the flows and volumes to the water quality model. WASP can be applied in one, two, or three dimensions. Hence it can be used to simulate the water quality impact of discharges into rivers, lakes, reservoirs, estuaries, coastal waters and sea.

DELFT3D

DELFT3D is a multi-dimensional (two- or three-dimensional) modelling system developed by Delft Hydraulics. It can be used to simulate the advection and dispersion of heat and chemicals discharged via cooling waters into fluvial, estuarine and coastal environments. It calculates non-steady flow and transport phenomena resulting from tidal and meteorological forcing. Areas of application of DEFT3D include salt intrusion, cooling water intakes, transport of dissolved materials and pollutants, tidal and wind driven flows, stratified and density driven flows. DELFT3D can simulate thermal discharge, effluent discharge and intake of cooling water at any location and depth.

DELFT3D can simulate a wide range of substances (such as chloride/salinity, oxygen, biochemical oxygen demand, excess temperature) and water quality processes (such as chemical processes, transport of dissolved substance towards the bottom, sedimentation and re-suspension of particulates).

Its non-hydrostatic assumption in the flow module allows for accurate predictions of vertical distributions of temperature and pollutant concentrations when the plumes are buoyant. This would produce more accurate predictions in the mixing zone.

TELEMAC 2D/3D

TELEMAC, developed by the Laboratoire National d'Hydraulique (a department of Electricité de France's Research and Development Division), is a modelling system for the simulation of physical processes associated with rivers, estuaries and coastal waters. TELEMAC comprises modules for hydrodynamics (TELEMAC-2D/TELEMAC-3D), water quality (WQ 2D/3D), sediment transport, dispersion of pollutants and wave dynamics. It represents a comprehensive range of water quality processes. TELEMAC can be used for dispersion studies of positively-buoyant (such as fresh or heated effluent discharged into the sea) or negatively-buoyant (such as cold or high-salinity effluent discharged into the sea) discharges.

THREETOX

THREETOX is a three-dimensional modelling system developed by the National Academy of Sciences of Ukraine for simulating hydrodynamics, salinity and thermal transport in lakes, reservoirs, estuaries and coastal ocean. It can also be used to model sediment and radionuclide transport, including the non-equilibrium partitioning of radionuclide in the water and sediment phases.

Key model input parameters

Model input parameters depend on the model used and predictions required from the model. Table 6-13 shows a checklist of the key input parameters for modelling the effects of thermal and TRO (from the use of biocides) discharges. Model predictions are expected to be sensitive to these parameters.

Table 6-13 Key model input parameters

Parameters	**Reasons for importance**	**Relevance**
Relating to the receiving water environment		
River cross sections and depths	Affecting flows	For river models.
Bathymetry of lakes, estuaries, coastal waters, sea	Affecting flows	For models of lakes, estuaries, coastal waters and seas.
Flows and tidal heights at model boundaries	Affecting flows	For models of estuaries, coastal waters and seas.
Non-physical model parameters		
Manning's roughness coefficient	Affecting flows	Manning's coefficient represents the resistance to flows in open channels and floodplains. Its value depends on many factors, including the bottom roughness/resistance, stage of flows, flow velocity, meandering of the channel and vegetation. Typical values range from 0.02 (e.g. for straight uniform channels with clean beds) to 0.1 (e.g. for natural streams with very weedy reaches, flood plains with heavy stand of timber).
Eddy viscosity coefficient	Affecting the vertical and/or horizontal mixing of mass and momentum.	Used in some 3-D models (e.g. DELFT3D). It is a factor for modelling turbulence. Typical values of the vertical eddy viscosity coefficient range from 10^{-5} m^2s^{-1} to 10^{-3} m^2s^{-1}. Values of the horizontal eddy viscosity coefficient range

Parameters	Reasons for importance	Relevance
		from 1 m^2s^{-1} (coastal areas) to 100 m^2s^{-1} (oceans).
Grid resolution and choice of modelling grid (e.g. rectilinear versus curvilinear grid)	Grid resolution and how the topography/physical environment of the discharge is represented affect the accuracy of the predicted water levels, flows and pollutant concentrations. In models that solve the hydrodynamic equations numerically, the grid resolution, together with time step, would affect the stability and accuracy of the numerical solution.	Horizontal grids need to be aligned with river channels and bends. For a 3-D model, typically a horizontal resolution of 10 m by 10 m (may be smaller if curvilinear grid is used) and a sigma coordinate of 10 to 20 layers in the vertical dimension for areas close to the water intake and outlet.
Size of modelling domain	It needs to be large enough for boundary conditions not to have any significant effect on model predictions within area of interest (particularly important for estuaries and coastal areas).	For models that solve the hydrodynamic and water quality equations numerically.
Time step	In models that solve the hydrodynamic equations numerically, the time step, together with grid resolution, would affect the stability and accuracy of the numerical solution.	For models that simulate the time variations of flows and pollutant concentrations explicitly. Typical values are in seconds and minutes.
Parameters related to the discharge		
Water intake: location, volume of water abstracted	These parameters affect the flows of the receiving water between the inlet and outlet locations.	For near and far-field models.
Water outlet: location, volume, velocities, temperature and chloride content of water discharged	Flow characteristics affect the advection and dispersion of discharged water quality determinand (temperature or chloride). The impact of the thermal and chloride discharges is directly proportional to the amount discharged.	For near and far-field models.
How results from a mixing zone study are represented as inputs	Inaccurate modelling results for the near field can affect predictions in the far-field.	For buoyant plumes and models that do not use non-hydrostatic pressure assumptions to model flow.
Parameters relating to model scenarios		
Seasonal variability in temperature, flows and concentrations of water	Affecting predicted flows and pollutant concentrations.	For near and far-field models.

Parameters	Reasons for importance	Relevance
quality parameters, ambient temperatures and wind speeds		
Tidal variations	Affecting flows and water depths.	For models of estuaries, coastal waters and seas.
Receiving water: temperature and chloride concentrations	They are the background temperature and chloride concentrations for the discharges to be mixed with.	For near and far-field models.
Meteorological conditions (wind speed and direction, ambient air temperature)	Affecting heat loss and producing wind driven current.	For models that can simulate heat loss and wind driven current.

6.3.5 Thermal preference and tolerance of biota

Recent reviews in the context of WFD requirements have been prepared on behalf of the Environment Agency (Turnpenny, Coughlan and Liney, 2006) and UKTAG (Turnpenny and Liney, 2006). Figure 6.26 provides a summary of thermal preference/tolerance data for a variety of fish species in relation to existing Freshwater Fish Directive and draft WFD temperature standards. Langford (1990) provides a useful summary of thermal effects, specifically relating to power station discharges.

Thermal tolerance depends on a number of factors, including zoogeographic origins of species and habit. In the UK, species originating from warmer climates south of the British Isles towards the Mediterranean are referred to as “Lusitanian”, while those distributed primarily north of Britain are known as “Boreal” or (further north) “Arctic-Boreal” species; these are sometimes referred to as ‘warm-water’ and ‘cold-water’ species respectively. Such distinctions are not absolute, and different species together occupy a continuum from north to south. They do, however, provide useful shorthand which indicates the likely responses to thermally enriched waters. Thus, bass (*Dicentrarchus labrax*) is a Lusitanian species which has its northern limits around the British Isles and benefits from the warmer waters around thermal discharges. Several decades ago, bass were scarce on the east coast of Britain and the northern limit of their distribution was around Sizewell (Suffolk). The thermal discharge canal at Kingsnorth was considered an important overwintering area that allowed bass to remain further north than they otherwise would (Langford, 1983, 1987, 1990; Pawson and Eaton, 1999). With gradually increasing winter temperatures over this period, bass have spread northwards and are now regularly caught by anglers on the east coast of Scotland[33]. While water temperature is a factor, Langford (1990) ascribes the attractiveness of these areas to bass as much to the food supply provided by entrained fish juveniles issuing from the CW discharge. The combined effect is that young bass not only survive the winter better in heated discharges, they also benefit from faster growth and a prolonged growth season. Pawson and Eaton (1999) argue that these benefits more than compensate for losses of young bass to impingement on the CW intake screens at Kingsnorth, which amounted to around 15 per cent of the available population in the autumn/winter of 1987 and 1988. So important are thermal discharges to juvenile bass in the UK that in 1990, MAFF established protected nursery areas around the larger estuarine and coastal power station thermal discharges in England and Wales[34].

[33] See e.g. www.worldseafishing.com/forums/archive/index.php?t-2203.html (viewed 10/02/09)
[34] The Bass (Specified Sea Areas) (Prohibition of Fishing) Order 1990: SI 1990 No.1156

Certain economically important shellfish are also of Lusitanian origin. Langford (1990) cites a number of examples of UK shellfish benefiting from, or at least thriving alongside, thermal discharges. The native European oyster (*Ostrea edulis)* favours warmer waters. Optimum feeding and growth in *O. edulis* occurs at a water temperature of about 20-25^{o}C, which is close to the upper ambient seawater temperature found in British coastal waters (Buxton *et al.* 1981). During the 1970s, its largest UK fishery was in the Solent, alongside the outfall from Fawley power station when it was operating at base-load. The Solent stock subsequently declined as power plant use became infrequent, the collapse due principally to overfishing. Similarly, the American hard-shell clam (*Mercenaria mercenaria*) thrived in the vicinity of the old Marchwood power station in Southampton Water, reaching unusually high population densities which then spread to areas outside the thermal field; following the closure of the plant, its stocks also declined, accelerated by overfishing. In these cases, biological detritus from entrained plankton may have contributed to the success of these stocks. Another interesting example more directly attributable to temperature is the improved survival of native oysters in the Blackwater Estuary next to the Bradwell thermal discharge observed after the exceptionally cold winter of 1962-3 (Figure 6-27).

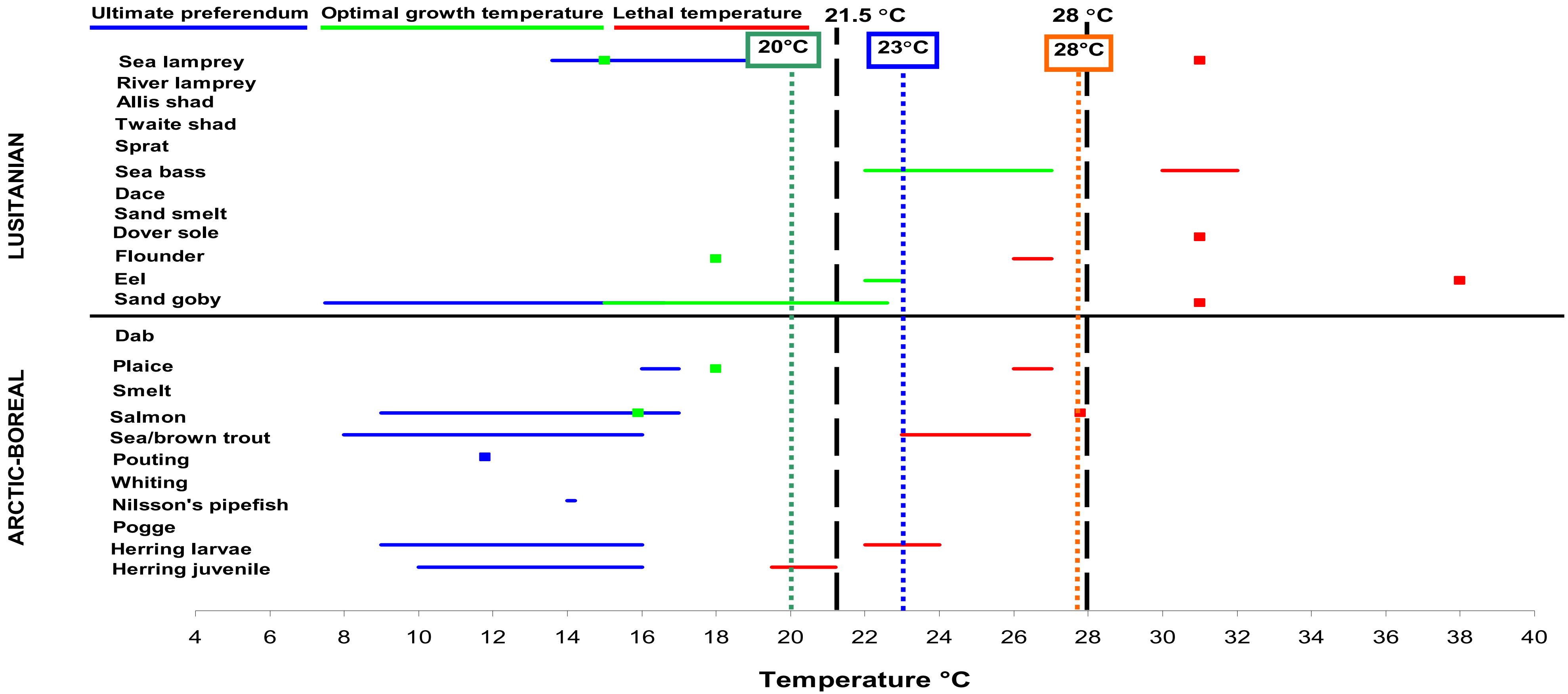


Figure 6-26 Temperature preferenda, optimal growth temperatures and lethal temperatures for key UK fish species. Current UK water temperature standards are indicated by dashed vertical black lines. Suggested WFD boundaries are shown by dashed vertical coloured lines: green, high/good; blue, good/mod; orange, mod/poor.

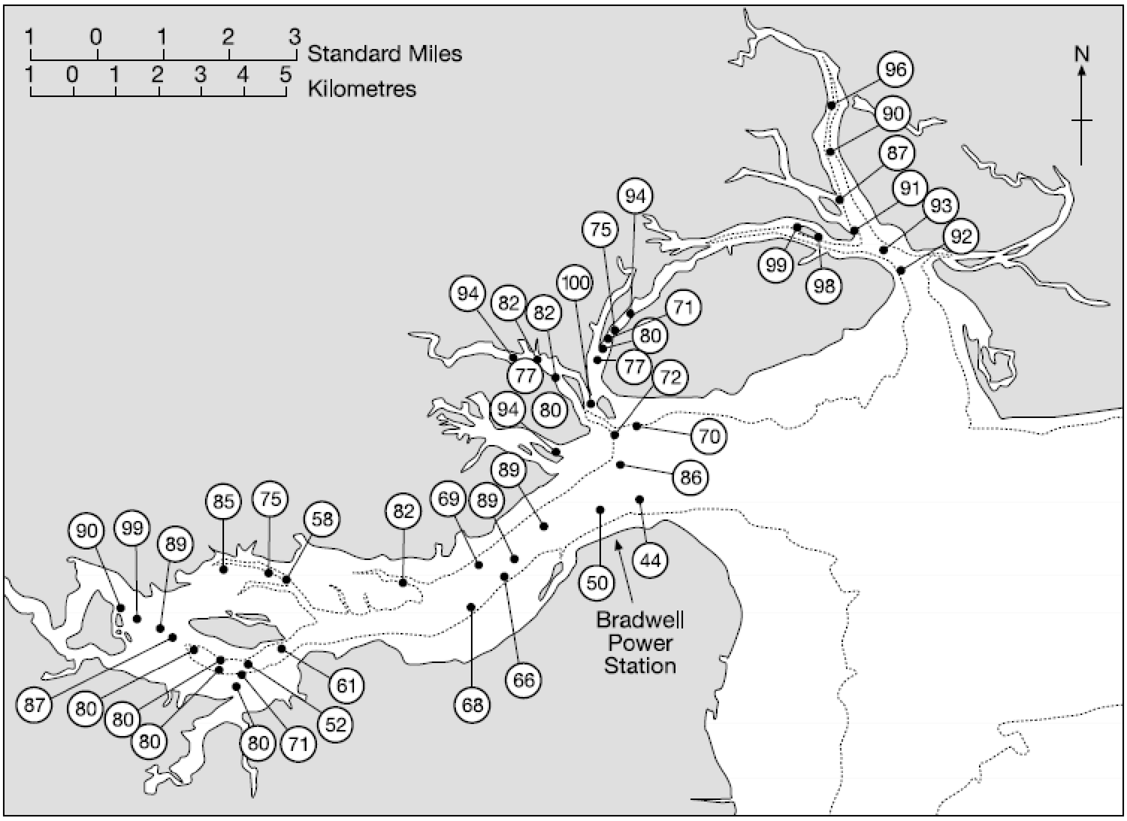


Figure 6-27 Mortality of oysters in the River Blackwater, Essex, following the severe winter of 1962-63 (Turnpenny and Coughlan, 2003)

The corollary of this zoogeographic effect is that species with more Arctic-Boreal origins which find themselves at the southern limits of their distribution in Britain may be excluded from heated areas. Members of the herring, cod and salmon families would fall into this category. However, the exclusion of a species from the relatively small area affected by a thermal discharge will in most cases be less ecologically significant than the foothold given to warm-water species by a thermal discharge. Problems could conceivably arise, however, if the plume or long-term thermal field affected a large part of a local habitat. Examples might include discharges affecting the benthic spawning grounds of herring (*Clupea harengus*), such as the Eagle Bank near to the Bradwell nuclear site, or the localised areas favoured by smelt (*Osmerus eperlanus*) in the middle zone of the Thames Estuary.

Species of intertidal habit are naturally adapted to wide fluctuations of temperature. Solar heat input affects the surface layers of all natural waters. In estuaries the heating and cooling of mudflats and other intertidal areas can cause strongly fluctuating temperatures on a tidal/diurnal timescale. The effects are most marked where intertidal areas form a high proportion of the total estuary area and in summer (this is particularly marked in estuaries where freshwater run-off and exchange is low and low water of high range tides occurs around midday). On clear, still summer nights significant re-radiation of heat can occur, at times resulting in local ground-frost. Spencer (1970) recorded a 15 °C variation in the near-surface temperature of a Milford Haven (Dyfed) mudflat over a 48-hour period in September 1968 but only 3°C in March of that year. In the Blackwater estuary (Essex) the heat rejected by Bradwell 300 MWe nuclear power station was equivalent to about 20 per cent of the incoming solar radiation on an average summer's day but about 200 per cent on an average winter's day. It was calculated that for every 3 °C through which the top 10 mm of mudflat was cooled by the returning tide, a quantity of heat equivalent to that rejected by the power station in one hour was rapidly transferred to the water (Hawes *et al.*1974).

The tolerance of an organism to a particular temperature regime will depend on the temperature to which it has become acclimatised. Thus, if an animal has adjusted to a low temperature regime and is transferred to higher temperatures, it will not cope as well as if it has had time to adapt its metabolic processes to the higher temperatures. This has limited significance for thermal discharges, as associated sudden temperature changes are seldom more than a few degrees Celsius; however, it is important to consider the relevant acclimation temperature when considering the applicability of temperature preference/tolerance results published from laboratory studies.

6.3.6 Effects of thermal discharges on marine benthos

Thermal discharges can only affect planktonic species as the effluent mixes with the receiving water and thus dilutes; as a consequence, such effects are negligible. Equally, nektonic/pelagic species, when not spatially constrained, can detect and avoid such effluent plumes if they are perceived to be deleterious. Conversely, the marine benthos is not as mobile as the nekton, and will receive the direct impact of the effluent where that water impinges on the sea bed or shore. It is thus the benthos which is most susceptible to effects of the thermal discharge. In practice, as the CW effluent water is normally less dense than the receiving water, it rises to the surface and away from the seabed where possible. Thus, any impact on the benthos is restricted to that area of direct impingement.

Species of the marine benthos will react differently to these aspects of the effluent regime, depending on their pre-adaptation to such conditions. In temperate waters in the northern hemisphere, species with a more southerly natural distribution will be more tolerant of generally higher temperatures; for example, species with a Lusitanian distribution will be more tolerant than those with a Boreal distribution. Species near the southern limit of their distribution (in the northern hemisphere) are more likely to be affected deleteriously by the thermal conditions of a CW effluent.

Species inhabiting the littoral zone are naturally exposed to greater ranges and extremes of temperature than those restricted to the sublittoral. While, when covered by the tide, the shore temperature will reflect that of the water, once uncovered exposure to insolation, wind and frost can lead to a radical and rapid change in temperature. During summer low tides in Milford Haven, Spencer (1970) measured temperature rises on the uncovered sand-flats of 0.2ºC min^{-1}, with a temperature range in September (1966) of 15 to 26ºC on the shore, compared with an ambient water temperature of 16.5ºC. On the mud-flats of the Medway Estuary (adjacent to the Thames) Walters (1977, in Bamber, 1990) measured an annual range from -4.5ºC to 32.5ºC.

The higher the shore zone, the longer the exposure to atmospheric (rather than aquatic) conditions, and thus the more extreme the conditions described above. As a result, the species which inhabit this zone are tolerant of (and adapted to) such temperature variations (within natural timescales).

Owing to the differential tolerance of species up the shore, the overall effect of these thermal stresses tends to be the moving down-shore of littoral zones, to the point of replacement of infralittoral and sublittoral species by those more tolerant species from the littoral (Bamber and Spencer, 1984).

There is also evidence that estuarine species, and those of coastal saline lagoons, are more tolerant of thermal stress than fully marine species (for example, *Idotea chelipes* versus *I. emarginata*: Naylor, 1965).

From the point of view of the benthos, three aspects of the temperature regime of a cooling-water discharge influence the biota:

1. Mean temperature in relation to "normal": the habitat local to a discharge will be warmer than the ambient conditions, by 8 to 12ºC adjacent to the discharge point and progressively decreasing in temperature differential (ΔT) further away as the discharge water cools.
2. Absolute temperature insofar as this can, in some circumstances, approach the thermal death point (upper incipient lethal temperature, UILT) of the exposed animals; where known, UILT values tend to be around 33ºC.
3. In tidal waters, there are tidal fluctuations of temperature at the seabed: where discharge water meets receiving water, there is normally a sharp interface or "temperature front" between these water bodies of different density; this front will be moved tidally across the seabed or shore on a regular cycle, and can lead to temperature changes of around 10ºC within 15 to 30 minutes (see Bamber and Spencer, 1984).

Mean temperature

The Lusitanian polychaete *Sabellaria alveolata* (the "honeycomb-reef worm") builds sand tubes in communal reefs. At Hinkley Point Power Station on the Bristol Channel, England, *S. alveolata* colonizes the shore around low-water (spring tide) mark, and develops larger reef units in the flow of the CW outfall water than anywhere else along the shoreline (Bamber and Irving, 1997). This species is constrained by low winter temperatures, water temperatures naturally falling below 5ºC in midwinter and air temperatures (during low tide emersion) below 0ºC, and indeed can be killed by frost when exposed at low tide. Reefs growing within the outflow experience winter water-temperatures no lower than 10ºC (on average above 13ºC, even in midwinter), protecting them from frosts and enabling them to continue growth during winter.

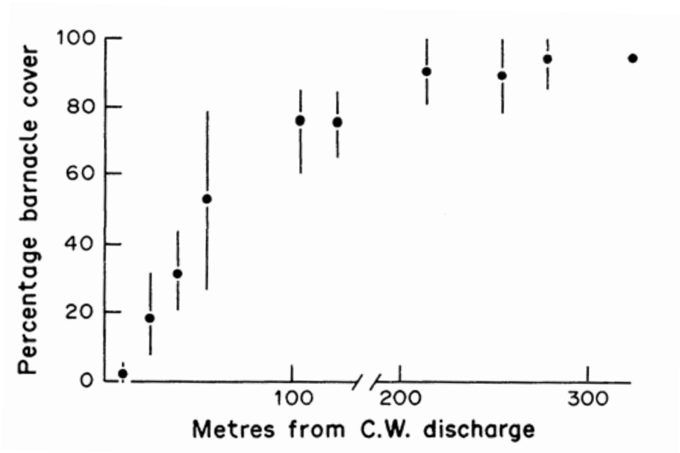


Figure 6-28 Percentage cover of *Balanus balanoides* (means with ranges; five replicates per station) for a transect of 10 sampling stations on the rocks leading away from Wylfa Power Station CW discharge (from Bamber, 1989).

In contrast, a survey of the northern rock barnacle, *Semibalanus balanoides*, on the rocky shore at the CW outfall of Wylfa Power Station (Bamber, 1989) found a significant decline in density with proximity to the discharge point (Figure 6-28). Such a gradient response is typical of a mean-temperature effect. A similar response was shown by sublittoral oligochaetes in the muddy benthos of Kingsnorth Power Station discharge canal (River Medway, Kent, England). A three-year survey of monthly sampling at six sites down the four-km length of the canal (Bamber and Spencer, 1984) found the common estuarine species *Tubificoides benedii* to show a gradient of decline from the control site (DC4) to the vicinity of the discharge headworks (N5), while, by contrast, the congeneric *T. amplivasatus* increased towards the headworks (Figure 6-29).

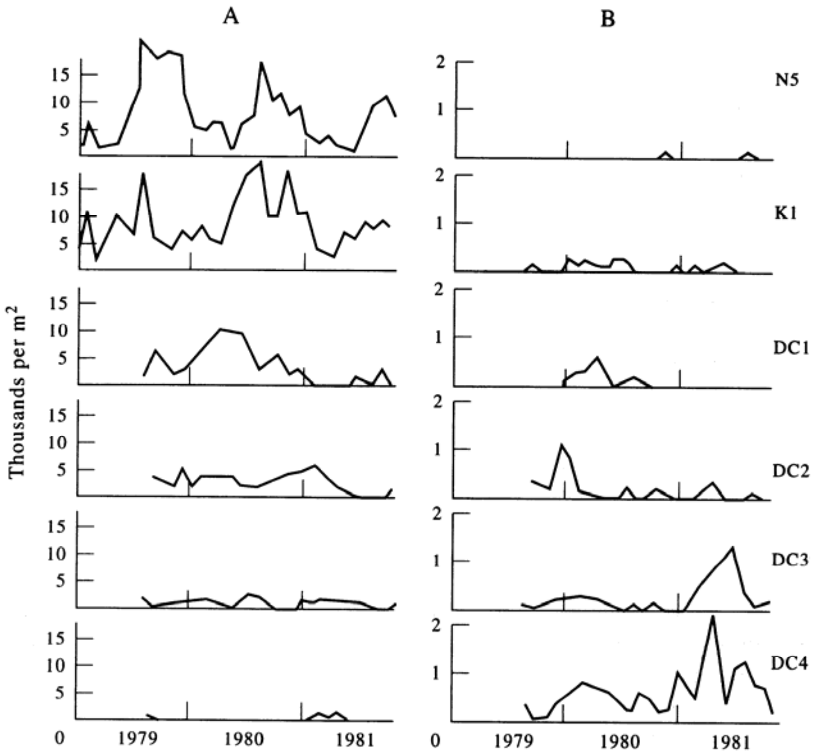


Figure 6-29 Monthly densities of A – *Tubificoides amplivasatus* and B – *T. benedii* in the CW discharge canal of Kingsnorth Power Station, at six sampling sites reflecting a gradient of mean ΔT from 9.2ºC at N5 to 0ºC at DC4 (after Bamber and Spencer, 1984)

A similar congeneric contrast by sympatric barnacle species in response to higher mean water temperature at a CW outfall was found by Straughan (1980a, b): on the shores at thermally affected sites in both Estero Bay and King Harbour, California, USA, the warmer-water, more southern species *Chthamalus fissus* was favoured over the more northern, cooler-water species *C. dalli*.

Absolute temperature

Examples of absolute temperature effect are less easy to detect, as they will commonly be expressed as an absence of a species, and thus less clearly detectable. During the study of the Kingsnorth Power Station outfall benthos mentioned above (Bamber and Spencer, 1984), one of the more abundant species of the littoral and infralittoral mud was the amphipod *Corophium volutator*. This species has been shown to become hyperactive at higher temperatures, and will leave its tubes to swim upwards into the water column at temperatures over 25ºC (Gonzales and Yevitch, 1977). In spring and early summer, the population of *Corophium volutator* near the Kingsnorth outfall headworks achieved densities of the order of 25 x 10^4 individuals per m^2; numbers then crashed as summer progressed and water temperatures rose above 27ºC, and the species was absent by August (recruiting again the following autumn/winter).

UILT levels have not commonly been calculated, but are generally around 30 to 33ºC (regardless of latitude) (see Bamber, 1990); summarizing the lethal temperatures for a large number of invertebrate species. Welch and Lindell (1980) found the statistical mode for the group lay between 35 and 40ºC.

The effectiveness of UILT levels is given by the fact that this absolute temperature effect is used by some power stations in Europe and the USA for biofouling control: cooling water is temporarily recycled within the CW system to kill off fouling organisms. For example, La Spezia Power Station, Italy, recycles water at 35ºC for 10 hours to control successfully fouling by serpulid polychaetes, barnacles and mussels (Jenner *et al.* 1998). At Eems Power Station, Netherlands, application of water at 38ºC for 30 minutes was sufficient to kill the barnacle *Balanus crenatus* and the mussel *Mytilus edulis*.

Tidal temperature-fronts

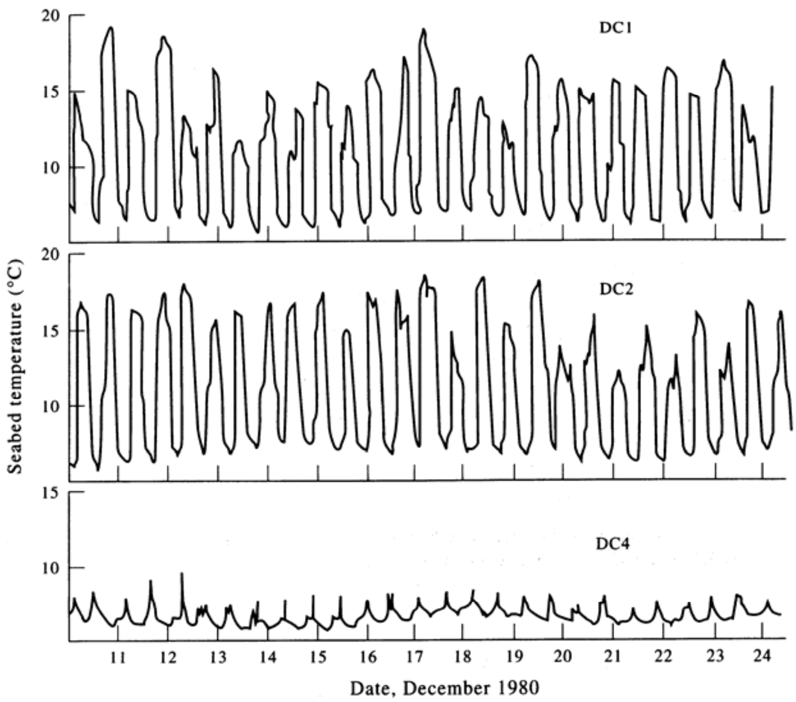


Figure 6-30 Continuous seabed temperatures in Kingsnorth Power Station CW discharge canal over two weeks in December 1980, starting at high tide, for sampling sites DC1 (one km from the outfall), DC2 (1.7 km) and the control site DC4 (four km)

The rapid movement of a temperature-front across the seabed represents a greater stress than the raised mean temperature, although it is limited to a smaller area. The thermal interface from the effluent at Kingsnorth Power Station (see above) was monitored at five-minute intervals over 18 months by Bamber and Spencer (1984): the temperature-front progressed the entire length of the four-km outfall creek on a tidal cycle, causing a variation of up to 12ºC at the seabed within 15-30 minutes (Figure

6-30, DC1, DC2). At the control site (Figure 6-30, DC4) a residual front of 3ºC was found on the first flood of the recirculating tide.

As a result, the number of species present at each of four studied sites along the length of the creek was consistently half of that at the control site (Figure 6-31), irrespective of the gradient of mean temperature (see above). The species not eliminated by this interface stress were all species whose range included the littoral zone, species naturally more tolerant of temperature fluctuation.

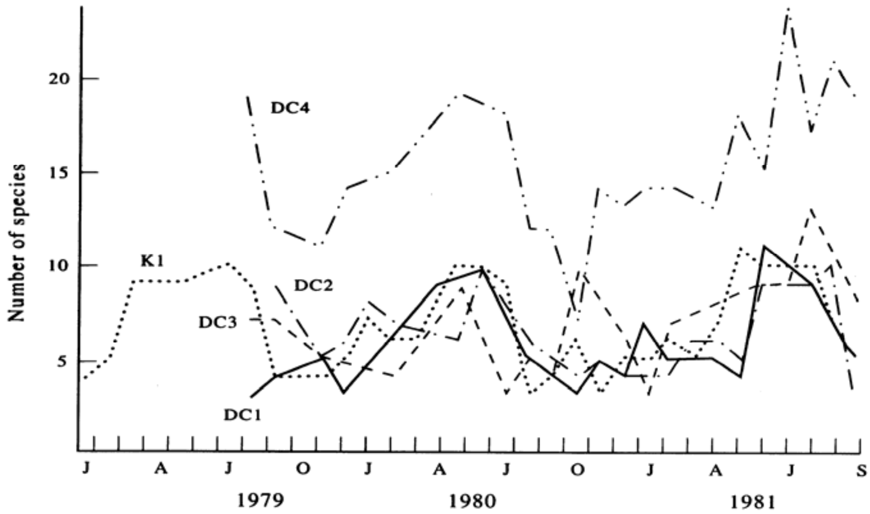


Figure 6-31 Numbers of benthic infaunal species in Kingsnorth Power Station CW discharge canal over three years; sampling sites K1 and DC1 to DC3 are within the canal, DC4 is the control site (after Bamber and Spencer, 1984)

Equally, the results from the control site indicate that a tidally-moving thermal-front of up to 3ºC was tolerable to these species.

Summary of effects on the benthos

The above examples show that warmer-water species are more tolerant of higher-temperature stresses than colder-water species, and that species whose distribution includes the littoral zone are more tolerant than those from the sublittoral. This is further demonstrated on rocky shores adjacent to thermal effluents by a downward shift of classical zonation (see Straughan, 1980a, b; Bamber and Coughlan, 1987).

The interrelationship between this evolutionary history, potentially resulting in some preadaptation to thermal effluent temperature conditions, and the three main stress parameters of those conditions (mean, absolute, and fluctuation in temperature) will govern the response of marine organisms to power-station CW discharges. Other than at outfall systems of a semi-enclosed configuration, thermal effects can be found within 200 to 500 m of the discharge, with some subtle effects a little further afield. Within this affected range, it can be predicted that eurythermal species from the littoral, from estuaries and coastal lagoons, and/or those from warmer biogeographic ranges will be unaffected or favoured until UILTs are reached, while stenothermal species from the sublittoral, fully marine and/or cooler biogeographic areas will be deleteriously affected. The extent and degree of any impact (deleterious or otherwise) will be proportional to the volume, velocity and ΔT of the discharged effluent, other things being equal.

6.3.7 Thermal barriers to fish migration?

The idea that thermal discharges, particularly when confined in estuary or river channels, can cause a barrier to fish migration is long-held but has rarely withstood scrutiny. Salmonids (including Atlantic salmon, *Salmo salar*, sea trout, *S.* trutta and European smelt, *Osmerus eperlanus*) are probably the migratory species of concern, as all three are Arctic-Boreal species (Figure 6.26), whereas lampreys (*Petromyzon marinus, Lampetra fluviatilis*), shads (*Alosa* spp.) and eels (*Anguilla anguilla*) are Lusitanian.

Turnpenny and Liney (2006) examined this issue in the context of developing WFD temperature standards for the UK and proposed that the allowable ΔT at the edge of the thermal plume should be raised to +3°C in place of the +1.5°C standard of the Freshwater Fish Directive, as scientific evidence shows that salmonids, considered to be the most sensitive indicators, are indifferent to crossing temperature increments of under 4°C, and will often cross ΔT-values several degrees above this. The +3°C uplift value has been adopted in draft standards, except in the case of high status waters.

A number of European countries place restrictions on the allowable thermal plume dimensions in estuary and river channels. This is to ensure that migratory fish are left a clear corridor within which suitable temperatures are maintained. While this approach was not adopted within draft UK WFD standards, it nevertheless seems a sensible precaution. Turnpenny and Liney (2006) recommended that the mixing zone should be contained within 25 per cent of the channel cross-sectional area for 95 per cent of the time. The five per cent time allowance recognises the uncontrollable spread of the plume in fluvial channels under slack water conditions and allows ample opportunity for migratory fish to pass. From a regulatory standpoint it provides a clear design criterion which can be tested against plume model outputs, although compliance is difficult to monitor.

Abundant evidence implicates temperature as one of the variables regulating the entry of salmon and sea trout from estuaries into freshwater. Studies on the Thames and other south coast English rivers studies have been reviewed by Turnpenny *et al* (2006), where more details can be found. The common finding is that a correlated group of variables (freshwater discharge/water temperature/dissolved oxygen) determines entry success into freshwater, rather than temperature per se. However, an important conclusion that can be drawn from such studies is that any temperature effect on salmonid migration will be exacerbated where DO is already depleted, for example by sewage inputs. In the Thames, data from Alabaster *et al*. (1991) suggest that a one mg l^{-1} fall in DO might be equivalent to a 4 °C rise in temperature with regard to influence on salmon return where ambient concentrations are below 5 mg DO l^{-1}.

Research has been conducted on the temperature sensitivity of juvenile fish that migrate along estuarine margins as part of their life cycle. The migration phenomenon is seen regularly during the summer months in estuaries such as the Thames, when a ‘ribbon’ of small fish such as dace (*Leuciscus leuciscus*), flounder (*Platichthys flesus*), elvers (*Anguilla anguilla*) and smelt (*Osmerus eperlanus*) occupies the shallow water margins of the channel (Naismith and Knights, 1988; Colclough *et al.* 2002). During migration they may be heading upstream or downstream, depending on species and habitat conditions. For example, in the case of elvers the migration is towards freshwater, where eels spend most of their life growing to maturity. For other species the purpose may be simply to disperse the population over the available habitat. The recently proposed consenting of a thermal discharge on the Thames Tideway has raised questions about whether intrusion of the plume into shallow marginal areas might cause a barrier to juvenile migrations, and this may have implications for any new nuclear stations constructed on estuaries. To resolve this, it is necessary to show: (a) that the plume does not impinge on the intertidal foreshore; (b) that the temperature rise is not sufficient to cause a barrier; or (c) that there are sufficient remaining migration paths or temporal windows of opportunity to ensure that the fish can pass.

RWE npower recently commissioned a series of experimental studies to investigate the tolerance of juveniles of selected fish species to crossing thermal interfaces (Jacobs, 2008). The experiments followed the design of Clough *et al*. (2000), in which fish were placed in an experimental channel with two parallel streams of water, one running at the ambient temperature (12^{o}C), the other at some higher temperature, to a maximum ΔT of +12^{o}C. The heated side was switched at each new temperature increment so that fish were encouraged to move if the temperature became uncomfortable. The results (Table 6.14) showed that of the five species tested, three (eel, goby and flounder) were indifferent to temperature increments of up to the maximum ΔT of +12^{o}C; only smelt (ΔT> +4^{o}C) and dace (ΔT> +8^{o}C) reacted at lower temperatures. These data provide a basis for assessing whether warm water from a thermal discharge entering the intertidal zone is likely to cause a barrier to migration of these species. All five species are commonly found in estuaries, where intertidal temperatures are naturally highly variable.

Table 6-14 Experimental data on thermal avoidance thresholds of juvenile estuarine fish (Jacobs, 2008)

Species (sample size)	Size (min-max) FL= fork length TL = total length	Temperature avoidance exhibited within test range (max ΔT=+12^{o}C)?	Base Temperature (ºC ± standard deviation)	Temperature at which avoidance is observed (ºC)	Temperature rise above base temperature at which avoidance is observed ΔT (ºC)
Smelt (n=48)	96-161 mm FL	Yes	12.29 (±0.28)	16	+4
Dace (n=60)	75-118 mm FL	Yes	12.23 (±0.24)	20	+8
Eel (n=60)	0.15-0.41 g	No	9.50 (±0.20)	-	-
Common Goby (n=60)	38-50 mm TL	No	12.32 (±0.25)	-	-
Flounder (n=48)	64-135 mm TL	No	12.37 (±0.25)	-	-

6.3.8 Phenology: alteration of seasonality

The thermal input to habitats exposed continually to a heated CW effluent will maintain an underlying higher temperature regime throughout the year, which can potentially influence behavioural and physiological processes in organisms living in that habitat. In temperate climates, this raised temperature can have the effect of altering the timing of seasonality as well as interfering with low-temperature physiology events.

Littoral populations of the sand-dwelling amphipod *Urothoe brevicornis* were studied by Barnett (1971) on the beach at Hunterston Power Station, Ayshire, Scotland, and at a neighbouring control site (three km away), comparing breeding and growth based on monthly samples for 17 months. The Hunterston animals (exposed to the CW effluent) showed an earlier onset of breeding, with earlier recruitment (by up to two months) and more prolonged growth than the control population (Figure 6-32). As a result, fully grown adults at Hunterston were some 20 per cent larger (linear measurement), although with no difference in growth rate.

The anthurid isopod *Cyathura carinata* is a European coastal species which has naturally spread northwards from a Mediterranean origin, now occurring as far north as the Baltic. Analyses of the population biology of this species from the Bay of Biscay, Kiel Canal and the Baltic demonstrate that, as this southern (warmer water) species has moved north, lower winter temperatures have constrained its life history pattern. Growth is reduced or absent through the winter in Baltic populations, such that most individuals are not large enough to reproduce at one-year old. There is thus a selection to increase longevity in northern populations, which acts to encourage progynous hermaphroditism (Bamber, 1985) and thus a highly biased sex ratio, largely or entirely female in the first year; eventually, in conditions where one-year-old maturity (1+ age-class) is minimal, hermaphroditism is no longer advantageous, and the sex ratio returns to 1:1. Thus, southern populations (such as at Arcachon: Figure 6-33) with good winter growth live only to 1+ with a 90 per cent maturity at one-year-old, early summer recruitment and a low female bias in 1+ sex ratio; northern populations (such as in the Polish Baltic: Figure 6-33) show no winter growth, live to 2+ with at most 50 per cent maturity at one-year-old, and have a high female bias in the 1+ sex ratio.

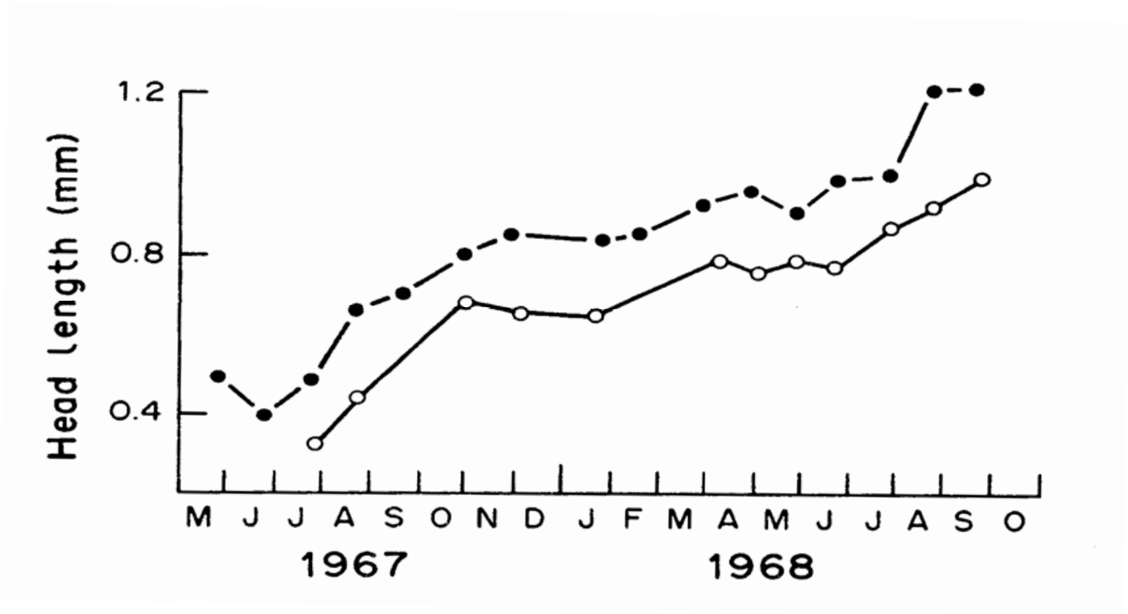


Figure 6-32 Growth curves for *Urothoe brevicornis* for the 1967 cohorts at Hunterston Power Station beach (solid circles) and the control beach at Millport (open circles) (redrawn after Barnett, 1971)

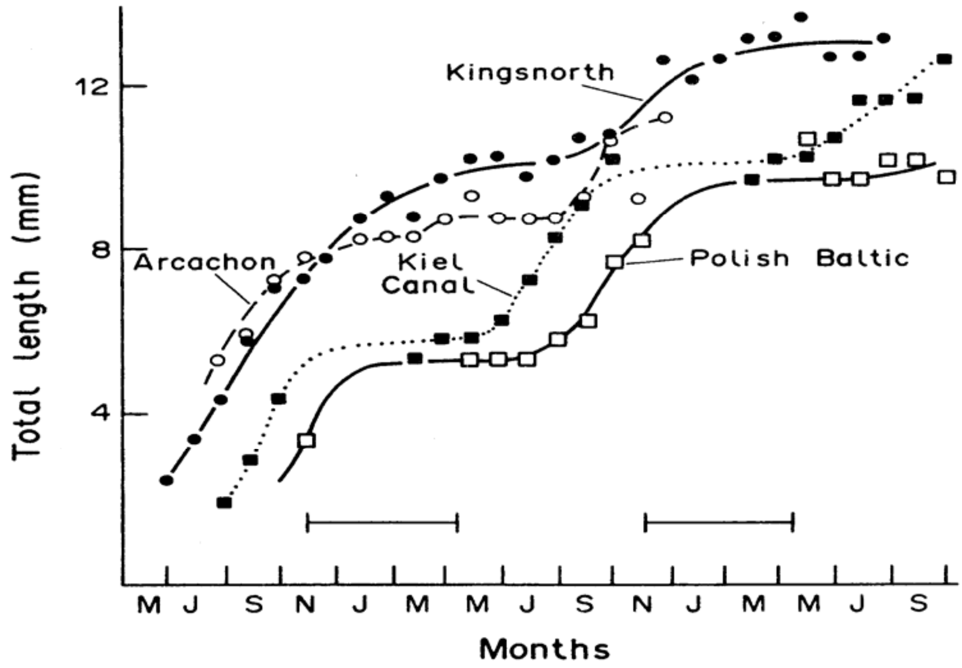


Figure 6-33 Growth curves for *Cyathura carinata* populations from Kingsnorth Power Station discharge canal (solid circles), compared with populations from Arcachon (open circles), the Kiel Canal (solid squares) and the Polish Baltic (open squares) (after Bamber, 1985); horizontal bars indicate winters

Bamber (1985) studied a population of *C. carinata* living within 100 m of the cooling water outfall headworks of Kingsnorth Power Station, on the estuary of the River Medway, Kent, England, where the water temperature was constantly between 8 and 10ºC warmer than the normal estuary temperature. As a genetically northern population, the Kingsnorth animals showed the 2+ longevity and highly biased sex ratio. However, with the winter temperature, enhanced by the CW effluent, not dropping below 13ºC, the population showed continuous winter growth (Figure 6-33), early summer recruitment in a successful population (the densest recorded from British waters) with over 90 per cent of the 1+ age-class being mature and breeding.

Conversely, the northern barnacle *Semibalanus balanoides* requires low winter temperatures for the onset of gametogenesis, and temperatures above 10ºC can delay fertilization indefinitely in this species (Crisp, in Naylor, 1965). While populations in the path of a CW discharge may be locally reduced in their density (see below), they will be affected further by an inhibition of their reproduction.

6.3.9 Effects of temperature on Natura 2000 sites

Provisional guidance on maximum allowable temperature rises for aquatic Natura 2000 sites was given in WQTAG160 by UKTAG (2006). Under this guidance the maximum allowable temperature uplift (ΔT) at the edge of the mixing zone is +2°C and the maximum temperature at the edge of the mixing zone is 28°C as a 98th percentile for a SPA or 21.5 ^{O}C for an SAC.

The guidance states that a mixing zone is acceptable if it can be shown that it has no effect on site integrity. There is no clear definition of exactly what affects site integrity, leaving WQTAG160 open to interpretation, although it does provide some guidelines:

- The mixing zone should not form a barrier to migration across the whole estuary or block areas of the estuary through which fish are likely to pass.
- In tidal waters, the discharge should not prolong the duration of the maximum natural temperature to the extent that it would begin to have a negative impact on biota (intertidal surfaces can reach temperatures above 30°C when exposed to summer sun, compared with water temperatures above 20°C in late summer).
- The impact of the mixing zone should be assessed in worst case conditions (such as low river flow and neap tide, when dilution of the plume is lowest).
- Additive spatial and temporal effects should be considered where more than one discharge impacts on the site.

These guidelines are now effectively superceded by draft WFD temperature standards, which would generate similar standards based on waters of high ecological status.

In addition to meeting standards on temperature, more detailed assessments may be required to ensure that each of the conservation objectives of the sites are met. This may mean, for example, considering the thermal sensitivities of species or biotopes listed in the site designation. Useful sources of information include UKTAG (2003) and Turnpenny *et al.* (2006) and Turnpenny and Liney (2006).

6.3.10 Design best practice and mitigation

A number of factors help control the potential impact of a thermal discharge on the receiving environment. The main ones are:

- size of the plant and its thermal efficiency, which dictate the rate of heat output;
- location of the CW outfall, which determines the proximity to any thermally sensitive habitats;
- design of the outfall structure, which determines how rapidly the heated water mixes and dilutes.

The various stages of design and assessment are discussed below.

Using water quality modelling outputs

Achieving the optimum balance for an environmentally acceptable solution may be an iterative process relying heavily on the modelling tools described in Section 6.3.3. Where existing nuclear sites are reused, it may also be possible to make use of historical data on the heat field and thermal plume behaviour (see Langford, 1990).

The process of optimising the outfall configuration makes use of habitat maps, usually stored as GIS layers, onto which graphical representations of the plume and long-term heat field can be overlayed. The GIS layers should include all potentially sensitive habitats, for example spawning and nursery grounds, fish migration routes, shellfish beds and any protected biotopes. Doing so identifies any possible risks of thermal exposure. This analysis is usually done with depth-averaged data displayed in plan view. Where the heat field is shown to overlie such habitats, model outputs are interrogated in more detail to determine the ΔT and time of exposure.

Within the field of the thermal plume, three-dimensional or layered data should be used, as depth-averaged data will not allow impacts on the bed versus the water column to be distinguished. An apparent high-temperature field shown in plan view may not in practice contact benthic communities. The long-term heat field is, by definition, fully mixed and two-dimensional data are therefore suitable for this purpose.

At specific locations, model outputs should be interrogated to generate exposure time-series. These show actual temperatures to which the habitats will be exposed and rates of fluctuation. A biological study is then required to assess the degree of impact.

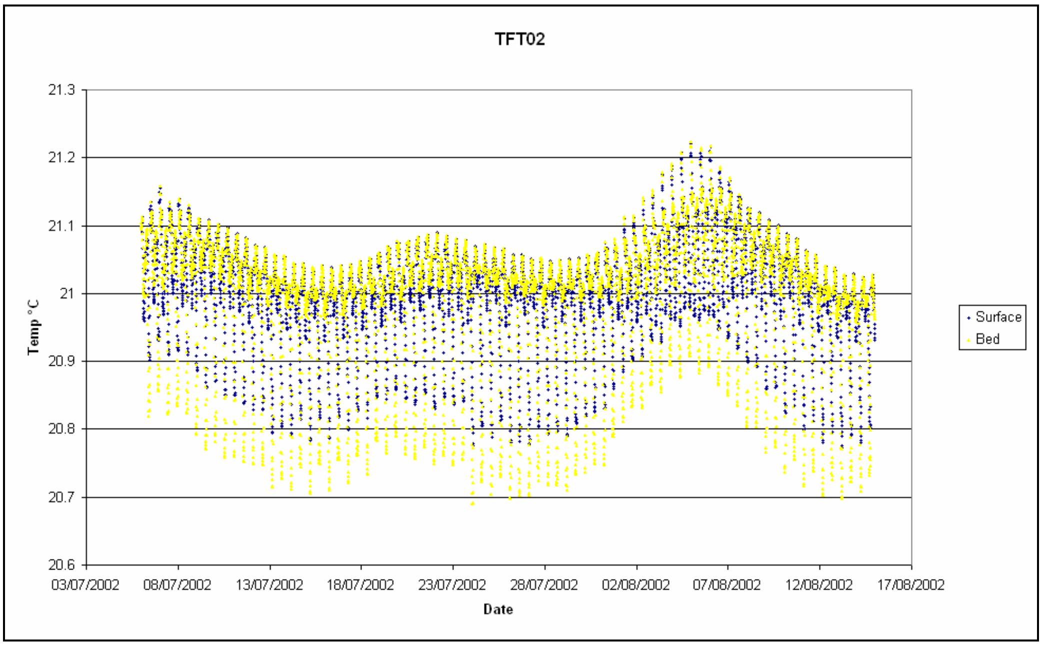


Figure 6-34 Example of a temperature exposure time-series for a specific habitat extracted from the Delft 3D model, comparing surface and bed temperatures

For estuarine channels or fish migration pathways, cross-sections representing different tidal conditions should be examined. The percentage of the cross-sectional area of the channel/migration path occupied by the plume at different tidal states should be calculated to ensure that there is a sufficient window of opportunity for migration (Figure 6-35).

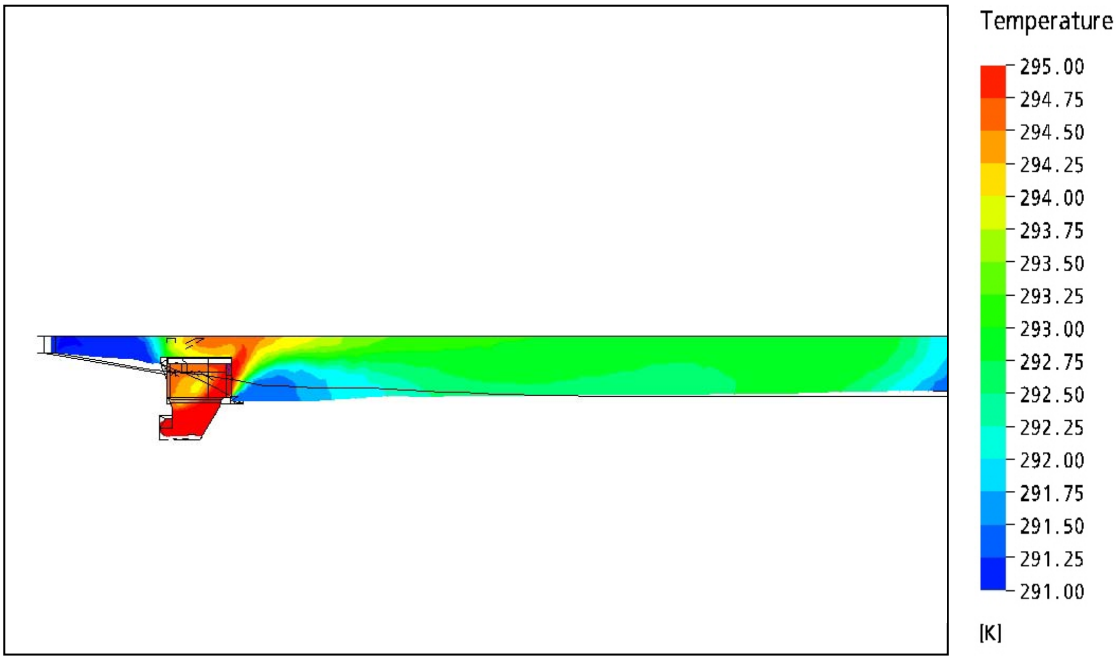


Figure 6-35 Modelled cross-section of a thermal plume entering an estuary channel, extracted from the CFX-5 model. This shows the buoyant plume rising from the outfall structure (to the left). Note that most of the channel cross-section contains ΔT values of under +3°C, which would be suitable for fish migration.

Figure 6-36 shows how models can be used to examine contact of the plume with the intertidal foreshore to establish potential risks to juvenile fish migration.

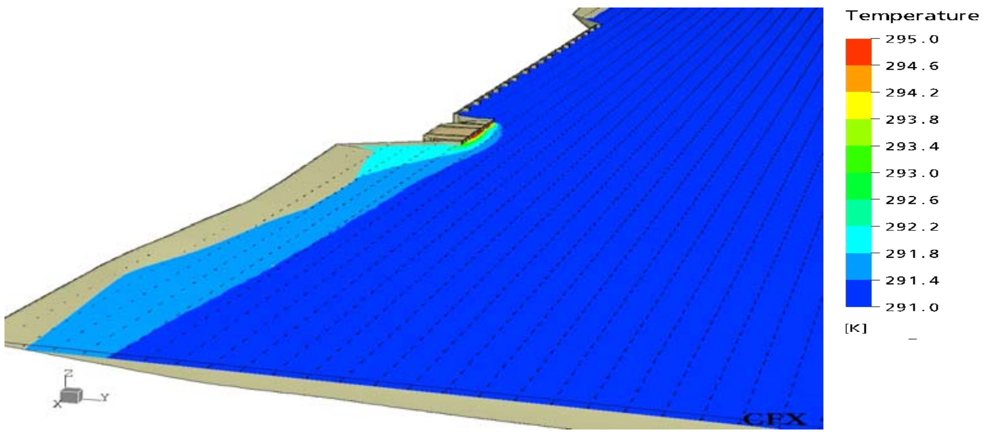


Figure 6-36 Isometric plot from CFX-5, showing the behaviour of the thermal plume, in this case clinging to the foreshore along the left-hand-side of the plot. This type of plot can be used to explore the risk of creating a barrier to juvenile fish movement along the intertidal foreshore

A similar process can be repeated for other water quality variables that may be altered by the discharge, for example, DO, ammonia and TRO.

Modifying the plume dispersion characteristics

Traditional outfall designs have relatively large ports that impart relatively little momentum to the water, leaving ambient velocities in the receiving water to disperse

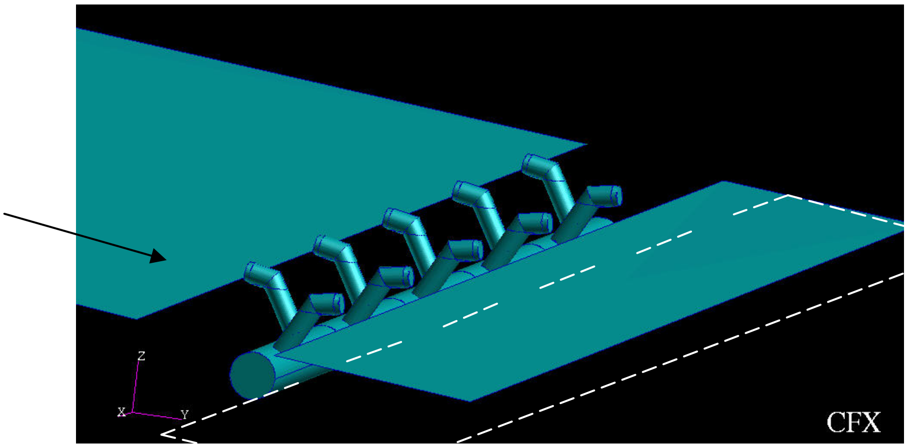


Figure 6-37 Depiction of a diffuser outfall with ten discharge ports. Arrow indicates tidal flow (reversible).

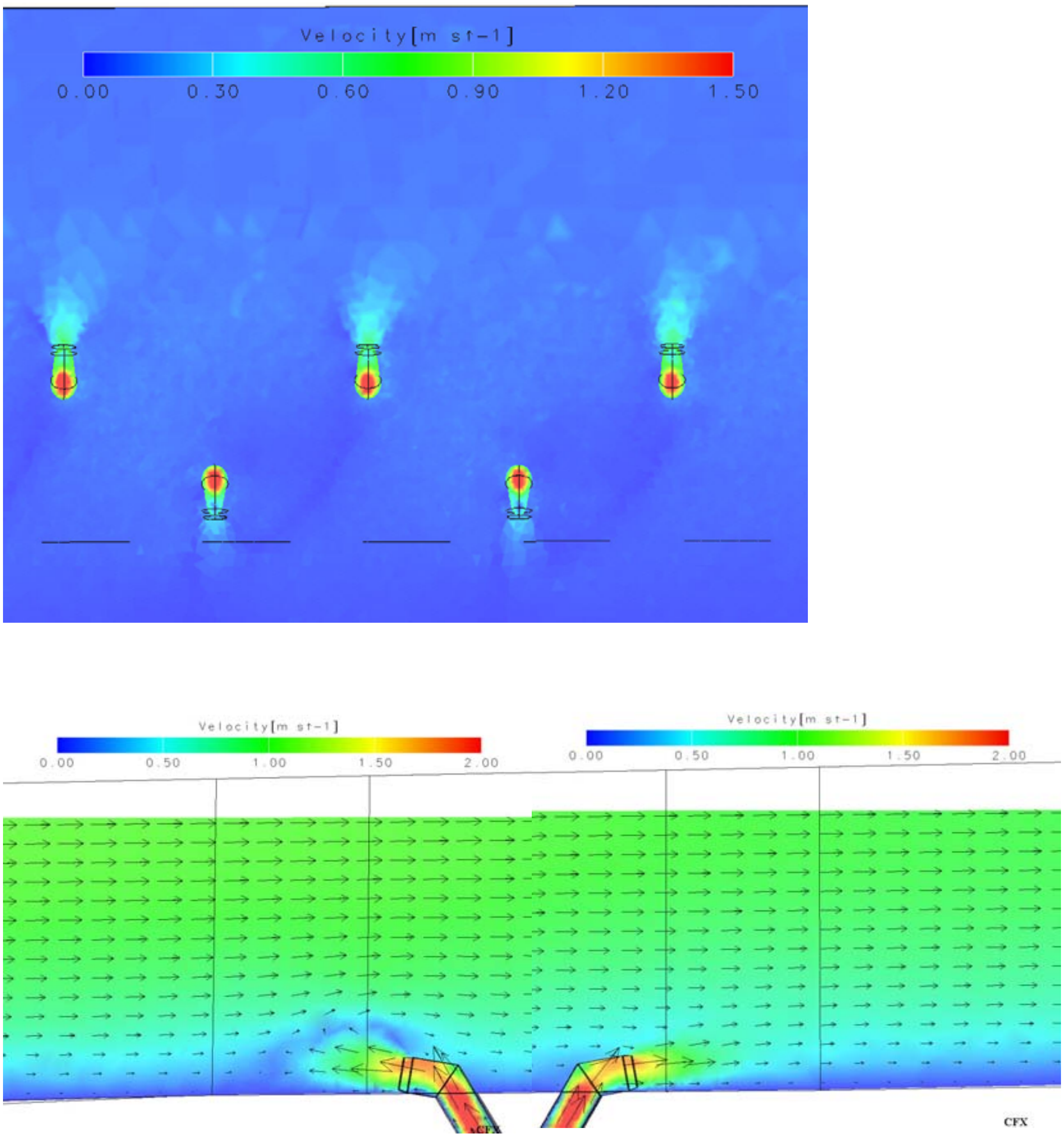


Figure 6-38 CFX-5™ model outputs depicting dispersion of heat from a multiport outfall. (Upper: plan; lower, vertical section)

the heated effluent. Various forms of diffuser can be fitted to the outfall to release the discharge over a distance, thus reducing the local concentration of heat. By forcing the water through narrow jets angled upwards, the plume can be forced towards the surface, yielding an almost two-dimensional plume of high temperature which is rapidly lost to the atmosphere (Shoener and Olmstead, 2003). Plume behaviour with different diffuser configurations can be tested by the US EPA-approved CORMIX model, or in more detail using newer CFD models such as CFX5 (Figure 6-38; see Section 6.3.4).

The mid-field and far-field are usually modelled using a three-dimensional hydrodynamic model driven by tidal level and currents at the model boundaries. This model should include all existing and future intakes and outfalls occurring within the long-term field, as well as the proposed CW system. Modern models use a triangular grid which can provide the fine resolution needed in the vicinity of the CW intake and outfall as well as the course resolution needed to cover the far-field. Typical output from a three-dimensional model includes plans and sections of the outfall plume at different states of the tide, and a plot of the intake temperature against time.

6.4 Using the cooling water stream to amend conventional discharges

The European BAT Reference on Large Combustion Plant (BREF LCP) states that BAT is to:

- prevent or reduce the amount of waste water;
- recycling of process and washing water;
- avoid direct contact of wastewater with the cooling system.

Nonetheless, as indicated in previous chapters, power stations do make use of the cooling system for rapid dilution of low-level radioactive waste and sewage treatment plant/grey water.

Other potential inputs to the cooling stream include desalination plant and flue-gas desulphurisation (FGD) effluents that use a seawater scrubbing process. FGD plant is only associated with fossil-fuelled power stations (such as Aberthaw). Seawater scrubbing can require an increase in the CW flow rate by up to 50 per cent, to provide adequate alkalinity for absorption of sulphur and nitrogen oxides.

Desalination plants built on new nuclear or conventional stations is a possibility in the future, to reduce reliance on borehole or town-main water. Quantities required for a 1,000 MWe plant or bigger are typically in the order of 3,000 to 6,000 m^3d^{-1} for steam raising and domestic use (Turnpenny and Coughlan, 2003). Assuming 50 per cent freshwater extraction, this would leave a similar daily volume to be discharged at twice the normal salinity. In, for instance, a 50 m^3s^{-1} direct cooling circuit, a dilution factor of between 700 and 1,500 would be provided by discharging into the CW stream.

7 Conclusions and recommendations

7.1 Environmental effects of cooling systems

7.1.1 Impingement

Fish impingement is minimal at inland sites with wet-tower cooling but can account for tens or even hundreds of tonnes of fish per annum at estuarine and coastal sites with direct cooling. Quantities of fish involved depend on the size and CW flow of the plant, proximity of intake to fish habitat and concentrations of fish and mitigation measures installed. The largest quantities recorded tend to comprise mainly pelagic fish such as sprat. At Sizewell A station, of 36 tonnes of fish impingement recorded in one year, 33 tonnes comprised sprat alone (Turnpenny *et al.* 1989). On occasions, sprat inundations at East Coast sites have amounted to hundreds of tonnes in a single day.

Although, these quantities appear large, they are small in relation to the industrial fisheries for sprat and of more interest are losses associated with impingement of juveniles of species of commercial or conservation interest. The removal of juveniles (undersized catch) by power stations has always been poorly regarded by the fishing industry and the Sizewell studies showed that for every tonne of juveniles removed by impingement, around fourteen tonnes of adults were in effect lost from production.

Mitigation techniques to reduce losses to impingement have progressed considerably in the past decade. At Shoreham CCGT station, which is fitted with an AFD system, the annual impingement rate averaged 3.8 kg per $10^6 m^3$, the lowest rate recorded for any UK station (see Figure 6-5). Combinations of techniques, including use of velocity caps (offshore intakes), AFD and FRR systems and other technologies that will be considered for the next generation of stations (LVSE intake designs, strobe light deterrents) should further reduce losses to impingement.

7.1.2 Entrainment

Entrainment affects all types of plankton but impacts on permanent zooplankton and phytoplankton (holoplankton) appear to be minimal in most cases, owing to the rapid turnover time of plankton populations.

Ichthyoplankton entrainment is potentially a more serious concern and has been extensively measured at US power plants under CWA s.316(b) rules, which require plant operators to assess entrainment impacts and put in place measures to reduce impacts by 60-90 per cent of unmitigated values[35].

Fish entrainment has been less thoroughly investigated than impingement at UK and European sites. The scarcity of entrainment data reflects a lack of regulatory interest in this less obvious form of impact. Entrainment impacts are potentially more significant than those associated with impingement, particularly now that impingement impacts can be largely mitigated.

[35] See e.g. http://www.powermag.com/water/Alternative-cooling-water-intake-analysis-under-CWA-Section-316b_1045.html (viewed 24/02/09).

The level of fish entrainment risk is determined by proximity to spawning grounds, and avoiding these and areas where fish eggs and larvae concentrate when locating the intake are the main means of minimising entrainment impacts.

The assumption of 100 per cent mortality of ichthyoplankton during cooling system passage advised by US EPA is not supported by UK power industry studies in which survivorship of entrainment has been measured using the EMU cooling system simulator. Assumed high mortalities associated with entrainment are a key factor in US EPA policy to limit direct cooling. However, UK data on survivorship are presently limited to a few species/lifestages. There is now a need for the industry to provide more comprehensive data covering vulnerable lifestages of species of commercial or conservation importance. It will also be necessary to demonstrate that the conditions tested effectively emulate those of new nuclear stations, which may entail conditions that differ from those already tested (based on the Sizewell B CW system profile).

7.1.3 Thermal discharges

Thermal discharge effects can be separated into near-field effects associated with the thermal plume and wider-range effects associated with the long-term heat field.

Water temperatures within the plume itself will typically be near or at the discharge temperature, which could be up to 10°C above ambient, depending on generating load and permitted ΔT value. Warm-water species may benefit from the plume and mobile species such as bass may move into the plume and benefit from food supplies, leading to faster growth and extended growing season. Shads and lampreys are also of Lusitanian origin (Figure 6-26) and are likely to benefit from warming in British waters. Conversely, some cold-water species will be eliminated from the swept area of the plume. These patterns are most evident in the extreme case of the Kingsnorth discharge, where the plume is confined within a tidal creek. This has provided excellent case studies of thermal effects (Section 6.3.6). Owing to its buoyancy, the warmer parts of the plume seldom have an impact on the benthos, except where the outfall is located onshore and warm water runs across the foreshore. Where this occurs, the plume may also cause scour until a stable bed form is created.

Where the plume impinges on the seabed in tidal waters, the discrete interface between the warmed effluent water and the cooler receiving water will move across the seabed: this rapid change in temperature represents the most severe thermal impact on the benthos, and will lead to a significant loss of species where the ΔT is above 3ºC. As littoral species are naturally more tolerant of thermal stress, the common result in the vicinity of thermal outflows is a down-shore movement (into the sublittoral) of the vertical zonation of benthic plants and animals, except where these are constrained by other environmental factors (such as light) when they are simply eliminated. These effects on the benthos are commonly restricted to within some 500 m of the outfall (depending on flow rate and volume, and physiography of the receiving water body).

A recent area of interest has been whether peripheral areas of the plume attaching to the foreshore of estuaries might inhibit longshore migrations of juvenile fish such as elver, flounder, goby, dace and smelt. New laboratory studies have shown that most of these species are indifferent to crossing temperature interfaces of up to a few degrees, with the possible exception of smelt. The smelt is a cold-water fish of the salmonid family and in the laboratory studies avoided ΔT values above +4°C. Where temperatures are expected to exceed avoidance thresholds, it will be necessary to ensure that this only happens for a small proportion of the tidal cycle, allowing sufficient window of opportunity for fish to migrate past.

Similarly, plume behaviour and temperature values in estuarine channels or at the mouth of an estuary must not be allowed to impede the free movement of migratory fish such as salmon and shad that use the larger part of the channel cross-section. Again, salmonids, being cold-water species, are the most sensitive to such effects but are not impeded by ΔT values under $+3^{o}C$.

The long-term heat field can have a number of effects, for example shifting zoogeographical limits of species already close to their northern (warm-water species) or southern (cold-water species) distribution. This occurs on a wider scale in the context of global climate change (and has always been a natural factor affecting marine pelagic fisheries) but for power station discharges has been seen most clearly in bass stocks on the east coasts of England and Scotland, where warming has improved overwintering prospects. Another interesting example was the enhanced survival of oysters in the River Blackwater around the Bradwell outfall during the severe 1962-63 winter. The opposite side of the coin is that cold-water species such as salmon and smelt which presently live close to their upper limits in southern British waters, may eventually be displaced by rising temperatures associated with climate change; where the long-term heat field of a power station coincides with a confined stock of such species, their displacement can be expected to accelerate.

The long-term heat field can also influence seasonality, through causing temperature-triggered biological events to occur earlier or later in the season. Higher temperatures usually mean an extended growth and reproductive season for warm-water species but may mean shorter seasons for cold-water species forced to live at temperatures above optimum. These effects are within the normal range of inter-annual variation for most species and will only be significant for climatic fringe species.

Other water quality effects associated with thermal discharges mainly arise from biocide residues. Chlorine, produced by electro-chlorination plants or derived from sodium hypochlorite, is by far the most widely used antifouling product and is used to prevent build-up of macrofouling (mainly mussels) and condenser slimes. It is normally injected at the CW intake to achieve an effective concentration of around 0.2 mg l^{-1} at the condenser inlet and will have declined to a fraction of this value through a series of reactions that take place during cooling system passage, normally allowing discharge consents to be met without the need for any form of dechlorination. After discharge, the concentration rapidly falls further so that there is no acute toxicity to fish or other species in the receiving water. During further stages of reaction with seawater, the residuals form a series of chlorination byproducts (CBPs), mainly organohalogens, which bond to sediments or in some cases can bioaccumulate in the food chain.

Dissolved oxygen concentrations in water are bound to temperature, first through solubility effects and secondly through temperature coefficients of metabolic processes. In most cases, effects of thermal plumes and long-term field on DO concentration will be insignificant but this conclusion may not hold in sewage-polluted estuaries where higher temperatures will drive up biological and chemical oxygen demand. Where this risk is apparent, water-quality modelling will allow an assessment to be made.

7.1.4 Cooling tower effects

The greatest public concerns over cooling towers are the visual impacts of the plumes on the landscape. Planning authorities are therefore reluctant to approve large, natural draught cooling towers. Other cooling tower types have a lower visual profile but take up large areas of land and are less energy-efficient.

Vapour plumes from wet cooling towers represent an aesthetic impact and can cause local fog formation. Ice formation on the ground during cold weather appears to be limited to areas in close proximity to the tower bases and is usually confined within the power station site. Similarly, salt deposition around seawater-cooled towers appears to be a minor localised effect, compared with natural salt deposition in coastal areas.

Legionnaires' disease, caused by the pathogen *Legionella pneumophila*, is the primary health concern associated with cooling towers. While outbreaks of Legionnaires' disease have been associated with air-conditioning towers, inevitably leading to concerns over the much larger cooling towers used on power stations, to date there has not been a single outbreak reported from this cause. Investigations have shown that, while the causative pathogen can exist within the lower parts of cooling towers, they have not been found above the eliminators or in the open air above cooling towers. However, it has not been proved that an outbreak could not occur. The risk is most likely to be associated with poor cooling tower maintenance (such as faulty eliminators) combined with atmospheric conditions favouring survival of the pathogen (dull, humid conditions).

7.1.5 Energy efficiency of different cooling systems

For reasons outlined in Chapter 2, wet (evaporative) cooling towers are more efficient in terms of delivering low recool temperatures, than dry towers and direct cooling is more efficient than tower cooling. Inter-station comparisons are of limited value since the major components and their design points will have been optimised for local conditions. However, it is now becoming standard practice to assess alternative cooling arrangements as part of the design and consenting process, comparing like with like. RWE NPower have compared direct cooling with two alternative methods for the steam turbines in the power-train of a notional 1000 MWe CCGT station (Mclauchlan, 2009). These data are show in Table 7-1.

The analysis in Table 7-1 was generated using modelling thermodynamic tools and selecting representative condenser back pressures for each plant design. It assumed a fixed boiler heat that would provide 1000MW net electrical output for a direct once-through cooling plant design. The Mechanical Draught and ACC designs were then run with this fixed heat input that therefore results in a lower gross generated electrical output due to the higher condenser back pressure. The total net electrical output reduction is a result of the lower generated gross output and auxiliary power requirements.

For the Mechanical Draught design, 78% of the net electrical output reduction is due to the lower gross generated output power with the remaining reduction coming from the increased auxiliary loading (combination of CW pumps and fan power). This is relative to the direct once-through cooling.

For the Air Cooled Condenser design 97.5% of the net electrical output reduction is due to the lower gross generated output power with the remaining reduction coming from the increased auxiliary loading (fan power). This is relative to the direct once through cooling and therefore highlights that the fan power is only slightly greater than cooling water pumps for the once through system.

Relative to the 1,000 MWe nominal rated output of the plant, these figures imply 2 to 2.3% lost electricity output.

Table 7-1 Potential loss of electrical output through the use of some alternative methods of cooling a 1,000 MWe power station, using direct cooling as the base case (Mclauchlan, 2009).

Parameter		Direct once-through cooling	Mechanical draught wet cooling tower	Closed-circuit air-cooled condensers
Condenser Design Pressure (mbar)		29	65	85
Reduction in electrical output (MW sent out)	MWe	0	42	47.4
	%	0	4.2	4.74
Reduction in Overall Cycle Net Thermal Efficiency (% points)		0	2	2.3

7.2 CW system design best practice

7.2.1 Selecting the most suitable system

General

Earlier sections of this report present the gamut of options available for power station cooling. Factors influencing the choice may include:

- sensitivity of source waters to abstraction impacts (entrainment and impingement), indirect cooling methods requiring less water and thus reducing these impacts;
- heat sink capacity of receiving water, lower capacities favouring indirect cooling methods;
- planning limitations on use of cooling towers (aesthetics, fog and so on);
- comparative lower thermal efficiencies of indirect cooling methods, therefore increasing carbon emissions per unit of electricity produced.

The first two of these favour indirect cooling, the second two, direct cooling.

Further factors that need to be considered are:

- whether predicted abstraction- and thermal-related impacts in the given situation exceed an acceptable level;
- whether the impact can be mitigated or compensated by way of, for example, replacement habitat.

Table 7-2 A comparison of cooling options

Environmental concern	Direct cooling	Cooling towers		
		Natural draught (wet)	**Mechanical draught (wet)**	**Natural draught (dry)***
Generation efficiency	High efficiency Uses less fuel so lower aerial emissions	Typically 0.5 - 1.5% less efficient than direct cooling	Typically ~2% less efficient than direct cooling	Lowest efficiency 2 - 3% less efficient than direct cooling
Complexity	Low	Moderate	High	Very high
Water abstraction	High	Moderate/low	Moderate/low	None
Abstraction effects	Site-specific -depends on characteristics of receiving waters			
Water consumption	None on-site	Moderate	Moderate	None
Visible plumes	None	Moderate	Moderate/low	None
Ground fog & icing	No icing. Local fog plume over shoreline discharges	None	Possible	None
Visual impact	Occasional foam or 'slick' at outfall	High	Moderate	High
Noise	None	Low	Moderate	Low/none
Discharge effects	Site-specific -depends on characteristics of receiving waters			
Waste disposal to landfill**	None if using fish recovery & return***	Moderate	Moderate	Moderate/none
Land use on-site****	None/low	Moderate/high	Moderate	High

* See sections 3.1.9 and 3.1.11

** Wastes from wet towers are mainly silt (non-hazardous); from dry towers, glycol (non-hazardous), if used

*** See section 7.2.3 'Consenting Issues' and section 6.1.6 'Biota recovery and return techniques'

**** This covers buildings and structures only and does not include spray ponds or cooling canals

Inland river or lake sites

It is less likely that new nuclear stations will be built inland but other large power stations will be sited inland and advice concerning these may be helpful.

Trawsfynydd nuclear plant, now undergoing decommissioning, is the only example of a lake-cooled plant in Britain and it is unlikely that thermal power plants will be built on lakes in the future.

Freshwater reaches of rivers are now considered unsuitable for direct-cooled stations and those, mainly coal-fired stations, constructed prior to the 1960s on large rivers such as the Trent have been phased out in favour of tower-cooled plant. All future river-cooled stations will therefore use tower cooling and where river cooling is not feasible, the option of dry-tower cooling remains open, subject to acceptable energy-efficiency.

The choice of which cooling system to use will depend mainly on planning issues. Natural draught towers are thermally efficient but are large and visually intrusive, particularly when creating vapour plumes. They rarely cause problems with local fogging or ground icing that may be found with lower profile mechanical draught towers. There have been no instances of *Legionella* outbreaks or other health problems associated with either type. Dry-cooling systems have none of these problems but are the least thermally efficient and therefore score badly on relative carbon emissions.

Estuarine sites

Direct cooling is still the most common method on Britain's estuaries, as large tidal fluxes provide efficient heat dispersal. However, estuarine locations are more sensitive than open coastal sites for three reasons. First, they represent important nursery grounds for many fish species such as sole and bass, and any impacts will have implications for the whole stock; secondly, they act as migration corridors for diadromous fish species, including salmon, sea trout, eel, shads, lampreys and smelt, as well as the young of various other fish species. Finally, estuaries are more sensitive to summer heat build-up, as flushing rates are lower than on the open coast and also large intertidal areas absorb solar radiation.

Mixed cooling is an option for estuarine sites. Barking power station on the tidal Thames runs for much of the year on direct cooling but is required under its consents to switch to tower cooling during the summer months when the river temperature exceeds 21.5^{o}C. The Doel nuclear plant, sited on the Scheldt estuary in Belgium, operates under a similar arrangement. Options include a full switch to indirect tower cooling at critical times, or bringing helper towers into circuits (Figure 4-15).

Two further factors affect the future viability of estuaries for direct cooling, particularly in the south of Britain. Background sea temperatures have risen over the past few decades and are expected under various climate change scenarios to continue to rise. UKCIP (2002) predictions suggest possible temperature rises in the southern North Sea and Eastern English Channel of between +2.5 and 4^{o}C (low to high greenhouse gas emissions scenarios) by 2080. Continuing background temperature rises are likely to restrict the ability of cold-water species such as salmon, sea trout and smelt to occupy southern estuaries. The second factor concerns organic (mainly sewage) pollution, which causes summer oxygen sag in many of our larger urbanised estuaries. It is usually a combination of low DO and high summer temperature that limits cold-water fish species (Turnpenny *et al.* 2006). Improvements in sewage treatment works to meet new WFD DO standards is expected to increase summer DO levels, which will have the effect of providing more thermal headroom for cold-water species; where this occurs, it will extend the future of cold-water species in southern estuaries.

Hard-and-fast rules about the type of cooling system to be used on estuaries are best avoided. Each case should be examined according to background temperatures, size of estuary, hydrographic conditions and environmental sensitivities.

Coastal sites

Tower cooling has not been used on any UK or European coastal site to date, direct cooling being the rule. Coastal sites do not suffer the same thermal capacity limits as estuaries, and provided that outfalls are carefully sited to avoid sensitive habitats heat disposal is not a problem. Given the better thermal efficiency of direct cooling, this would be the preferred option, provided that impacts associated with abstraction are acceptable.

7.2.2 Intake structure design and position

All intake types

The type of intake design will depend mainly on its position, which may be onshore, for example as an opening in a wharf (Marchwood, Kingsnorth), onshore via dredged channel (Fawley, Uskmouth, Oldbury), or offshore (most UK nuclear stations).

The primary consideration in selecting an intake position should be to avoid thermal recirculation. This would represent a major environmental impact through loss of thermal efficiency and therefore increased carbon emissions to replace lost efficiency. Other key concerns are to design an intake that is safe to humans and wildlife.

No formal human safety guidelines exist for intakes but the main requirements are that intakes should:

- not create a surface vortex that might endanger craft or swimmers;
- have entrances protected by bars to prevent entry;
- maintain velocities across the face of the bars that would not pin a swimmer or diver to the bars.

These criteria would also protect aquatic mammals. Bar spacing and velocity criteria selected for fish exclusion should be more than adequate for human safety.

Most important for protecting wildlife is to avoid proximity to important fish spawning and nursery grounds. How close is too close? The studies mentioned at Bradwell and Pembroke (Section 6.1.6) provide some insight, where in both cases herring spawning grounds were 10-12 km from the intake positions. In the case of Pembroke, field surveys established that entrainment was unlikely to exceed 0.1 per cent of the Milford Haven adult stock. The Bradwell study showed that up to a quarter of the available herring larvae and post-larvae might wash past the intake, but it was not established how many of these would enter the intake, although concentrations in the cooling water appeared to be only around 10 per cent of those in adjacent open water (Coughlan *et al.* 1980). However, it would be dangerous to apply any simple distance rule-of-thumb as the risk to stocks depends greatly on the local hydrographic conditions and where the intake lies relative to the drift path from the spawning ground. A single tidal excursion might carry spawn or larvae over 10 km or so. It is necessary, therefore, to undertake a hydrographic analysis similar to those described in Section 6.1.6, or to measure actual concentrations of ichthyoplankton in the abstraction field of the intake.

Different principles apply when considering proximity to nursery grounds, as by the time fish have graduated to nursery areas they are no longer planktonic, therefore less at the mercy of currents. More effective mitigation measures can be provided in the form of fish deflection and return systems. Less reliance will be placed on mitigation measures when the intake is located in an area naturally low in juvenile fish. This means avoiding as far as possible abstraction from intertidal areas, including estuarine foreshore, saltmarshes and the surf zone of beaches, except where it can be shown that the habitat is relatively unproductive. Further specific advice is given below.

Onshore intakes

Positioning

Onshore intakes that use dredged channels run the risk of entrapping fish from highly productive intertidal habitat, the fish tending to run into deeper channels towards low tide (Figure 6-12). In estuaries, intakes of this type are also likely to intercept juvenile fish that migrate along the channel margins. One option might be to sheet pile the channel sides so that intertidal drainage runs away from, rather than towards, the channel. Dredged onshore intake channels across saltmarshes or other high quality habitats are best avoided entirely.

Wharf-type onshore intakes that open directly into deep water are less of a risk to intertidal biota and are the preferred option where an onshore intake is used.

In either case, it is important to avoid locating the intake in a backwater area where large quantities of weed or other debris might accumulate. Where stations are to be built on existing power station sites, historical knowledge of screen blockage problems should be reviewed.

Design of structure

The main requirement is to ensure that intake openings are large enough to keep approach velocities below recommended values. Environment Agency Best Practice (Turnpenny and O'Keeffe, 2005) allows a flexible approach to setting intake velocities, using data on fish swimming speeds. This ensures that the intake structure is no larger than necessary, keeping down costs, the physical and carbon footprint and reducing sedimentation risk associated with low water velocities. In practice, entrance velocities at maximum abstraction rate normally fall between 0.3 and 0.5 ms^{-1}.

Fish deflection measures

AFD systems, strobe lights or other deflection devices are normally installed in or immediately in front of the intake openings. In rarer cases, they are mounted on specially constructed frames or piles placed some distance upstream of the structure where water velocities are lower (see Figure 6-19). For new plants it is preferable to design the intake openings to have suitable velocities in the first place. For AFD systems, acoustic modelling is an essential step in the design process to achieve an effective but confined sound field.

Offshore intakes

Positioning

Offshore intakes take abstractions away from sensitive intertidal nursery areas but the precautions listed above for spawning areas still apply. Locating an intake offshore may affect commercial fishers by interfering with trawl lines. It may in some cases increase the risk of taking in species migrating along the coast, such as salmon (for example, on the north-east coast of England, where T-net interception fisheries operate) or of sprat inundations (predominantly North Sea and Eastern English Channel).

Design of structure

Offshore intake structures used in the past have not usually been designed with fish protection in mind, or at least not with a good understanding of the subject. There is no excuse nowadays for building a structure that suffers from the problems highlighted in earlier sections of this report (6.1.6). The two main offences are (1) allowing tidal stream velocities to add to inlet velocities, and (2) designing a capped structure that does not conform to the hydraulic requirements for a velocity cap. The LVSE intake (Figure 6-17) provides one possible design but has not yet been developed and tested for nuclear power stations. Any design that can be shown in model tests (preferably with physical modelling) to meet the velocity requirements should be acceptable.

Biota deflection measures

As for onshore intakes.

7.2.3 CW screening and fish protection systems

Choice of screen type

Primary screening at the intake point using passive wedge-wire cylinder (PWWC) screens, although in theory scalable to handle tens of cumecs CW flow, is unlikely to be acceptable to nuclear operators owing to unproven viability in the hostile offshore conditions of the UK, blockage risk, and difficulties of access for maintenance.

Fine screening using band or drum screens can provide suitable protection for the main cooling circuit as well as for entrapped biota (using FRR techniques – see below). The preferred mesh size may depend on the developer, European stations traditionally using finer (2-3 mm) screens than British counterparts (6-10 mm). For FRR purposes, a screen size of up to six mm is advised but meshes this small or smaller at coastal sites may lead to a risk of ctenophore blockage during the summer months.

Use of intermediate raked bar screens may be required at some sites. Their use is not desirable from a fish protection point of view but, if used, consideration should be given to safe and efficient fish handling.

Fish deflection/fish recovery and return systems

Detailed advice on these subjects is given in the Environment Agency Best Practice Guide (Turnpenny and O'Keeffe, 2005) and only outline details are given here.

Where PWWC screens are not used, best practice for directly cooled coastal sites allows a combination of acoustic fish deterrents (AFDs) and fish recovery and return (FRR). The combination of both processes caters for hearing-sensitive, delicate species (AFD) as well as more insensitive demersal and epibenthic species, including crustaceans (FRR).

Although the best practice guidance is relatively recent, fish protection technology is moving fast, partly in attempt to keep up with developing European legislation (notably the eel management regulation and new fish pass and screens regulations proposed for the 2009 parliamentary session to bring controls up to Water Framework Directive standards. Consequently, it may be necessary to go beyond current best practice, with better provision to protect eels. This will require the development and trialling of larger fish buckets for band and drum screens, more suited to handling sinuous species such as eel and lamprey. Fish deterrent strobe lights also appear promising for eel deflection. Accordingly, the recently granted Abstraction Licence (Ref. No. 22/61/6/156) for Pembroke CCGT station includes a requirement to install and test the efficacy of strobe lights at the intake.

Some problems still need to be resolved with FRR systems. In the case of band screens, if larger fish buckets are to be used, openings through which the band screens pass may need to be enlarged to allow the larger buckets to pass through. For new systems built from scratch this should not be a problem. Where older facilities are being reused, as has been the practice at numerous recent new builds, openings may require modification. On standard UK drum screens, fish buckets are fitted inside the drum and this is not an issue. Hopper geometry on drum screens does, however, need to be checked (see Section 6.1.6).

Some difficulties have arisen in the application of best practice for FRR systems. One aspect concerns the requirement not to chlorinate ahead of the screenwells, to avoid exposing fish to the toxic effects of the biocide. While concentrations and exposure times may not be directly lethal, to use chlorine before this point is likely to disorientate fish and leave them more vulnerable to predation at the return point. Where there is a risk of biofouling upstream of the screens becoming an operational threat, plant designers/operators need to find effective solutions. These could for instance include periodic shock dosing of individual screenwells when offline, treatment of surfaces with fouling-resistant paints or gels or manual cleaning as required.

The advice that bends in launders from FRR systems should have a minimum radius of three metres has also been difficult to achieve in some cases, especially where old infrastructure has been reused with insufficient space to achieve this. The purpose of this criterion is to ensure that material does not become caught up in bends, restricting water flow and trapping fish. Where this cannot be achieved, the developer should be able to demonstrate that this will not happen, for example by using larger channel sections and ensuring adequate water flow.

The predation rate at FRR outfalls has received little attention to date, even though experience in similar situations suggests that predators are attracted to any discharge containing potential food items. This aspect would merit further consideration.

In-plant monitoring facilities

Abstraction licence conditions for recently consented power stations have included a requirement to monitor impingement and entrainment rates. Specifying sampling points at the design stage facilitates this.

Entrainment sampling

Sampling on older plants has normally required dropping plankton pump samplers or plankton nets into the CW forebay. The procedure is unwieldy and time-consuming and, where tethered nets are used, turbulence makes quantification difficult.

A better option is to draw water from the CW culverts via a tapping on the pressure side of one of the CW pumps. At this point, any young fish will be in the dark and unlikely to orientate to avoid entrainment into the sampling offtake. The sampling offtake should use at least a 100-mm inside diameter tapping and be capable of delivering at least 25 $l\ s^{-1}$ flow. It may be worth installing, say, three tappings side by side so that sampling can be done in triplicate.

A standard pipe tapping would draw water primarily from the boundary layer of the CW culvert, which may not be ideal. The alternative would require the sampling point to project at least some distance from the culvert wall. No design was found for such an arrangement but a simple hydraulic analysis, for example using CFD modelling, would enable the development of a suitable design.

In some cases entrainment sampling points have been taken from screen washwater pump supplies. This arrangement is second best as the much smaller size of the pumps will mean that a large proportion of the ichthyoplankton is likely to become damaged, making identification difficult.

The sample is collected by filtering the sample flow through a plankton net suspended in water, normally for a 24-hour period (though maybe split into shorter subsampling periods). The facility must therefore incorporate some sort of tank and a drain. A suitable arrangement is shown in Figure 7-1**.**

Impingement sampling

The arrangement in Figure 7-1 also caters for impingement sampling. Material normally returned to source via the FRR launder is diverted into a basket in a collection well. Water level in the well is maintained by a weir. The set-up is similar to many trash basket arrangements at power stations, except that a weir is used to maintain a water level. On new nuclear stations, owing to CW requirements, the amount of material passing through the system will be large at times and sampling systems must be able to cope with this. If there are four screens it may be necessary to have four separate sampling wells, or one shared between two screens. Lifting gantries will be required.

To reduce the risk of fish exhaustion or damage during retention in the collection well, the design should ensure that energy dissipation within the well is kept at or below 100 Wm^{-3} (see 6.1.6).

Consenting issues

Discharge of FRR systems to sea

There has been some confusion in the past about the status of backwashings from CW screens and whether their discharge to sea should be treated as a trade waste. The precedent of returning backwashings to sea was established for Sizewell B, owing to the perceived ecological benefits of returning both live and dead material to the ecosystem (Turnpenny and Taylor, 2000). A number of new stations have followed suit. The Environment Agency has no formal policy, but the following may be helpful:

- If the waste is returned to the water in a continuous stream, along with entrained material, there should be no consenting issues. This reduces landfill waste.
- Maceration of screen arising may be acceptable if discharged continuously. A degree of maceration occurs incidentally on some current, more primitive fish return systems.
- Concentrated dumping of accumulated waste would be a different matter and could have significant waste and water quality issues.

On the last point, consenting of the Sizewell B fish return system required a means of monitoring fish throughput in the return system (visual observation/CCTV camera) and that the fish return launders should be diverted into trash baskets if abnormally high numbers of fish were being returned to sea. The provision was made owing to previous experience at Sizewell A of sprat inundations, and concerns that large numbers of dead fish might wash up on local beaches. As part of the commissioning phase studies at Sizewell B, over a period of six weeks releases of thousands of dye-marked dead sprats into the CW discharge were made. During this period, National Rivers Authority staff monitored Sizewell Beach but recorded no incidence of marked sprats being found on the beach, and no higher incidence of dead fish than when the plant had not been operating (Turnpenny and Taylor, 2000).

All of the above indicates a need for case-by-case consideration. The issue of reducing fish kill will influence future policy and practice over biodegradable waste of this origin.

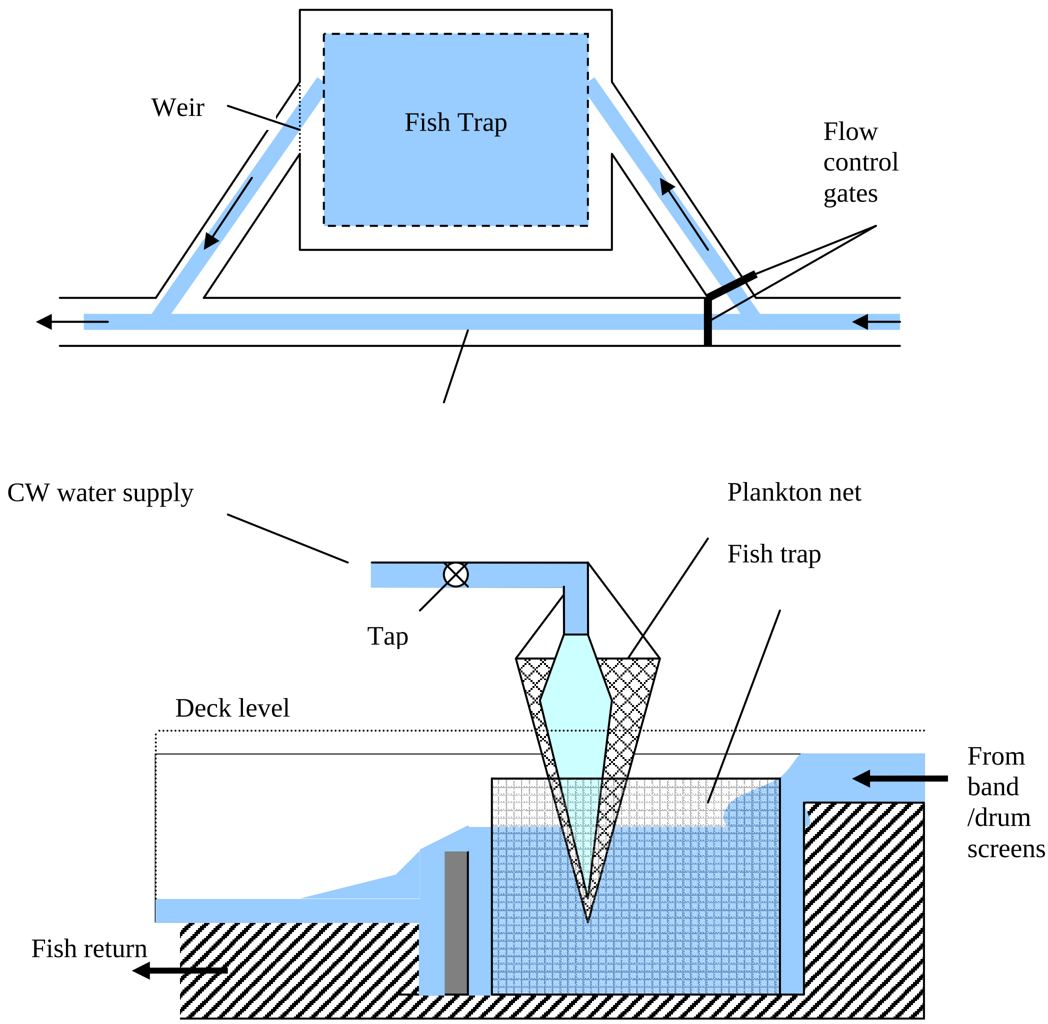


Figure 7-1 Example of sampling facilities for impingement and entrainment monitoring. Upper drawing shows diversion from FRR launder into water-retaining collection basket for impingement sampling. During sampling, shutters or gates are adjusted to divert flow into sampling well. Lower drawing shows the arrangement in vertical section, along with the suspended plankton net used for entrainment sampling.

Trialling mitigation techniques

Useful development of improved biota protection techniques can often be achieved in laboratory flumes or other systems but trialling necessarily involves testing on live plant. Fish deflection systems are normally tested by comparing impingement rates over alternating on-off periods. Excessive regulation can prove a hindrance to this process. For example, the introduction of combined AFD and FRR techniques at

Shoreham CCGT power station in the 1990s led to the lowest recorded rate of fish impingement for any UK station investigated up to that time (Clough *et al.* 2003) but the true efficiency of the AFD system could not be investigated as the abstraction licence required the mitigation measures to run at all times when the plant was operating. In more recent abstraction licence negotiations, it has been agreed that this requirement can be relaxed for the purpose of scientific testing of mitigation measures. This approach could usefully be adopted at other new build sites.

7.2.4 Cooling towers

It is unlikely that nuclear station developers would opt for indirect cooling by choice, owing to the various penalties: planning issues, land-take, capital and running costs, lower efficiencies, waste-disposal and so on. If forced by aquatic environmental issues to use indirect cooling, tall natural draught towers in the coastal landscape are unlikely to be acceptable to planners for aesthetic reasons, whilst dry cooling or condensing would be an unlikely choice on cost and efficiency grounds and is only defined as BAT where water supply is limiting. This leaves large arrays ("streets") of mechanical draught towers. Since the most likely nuclear sites are coastal or estuarine, the towers would be using seawater and so in addition to planning, cost and efficiency, there would be issues of drift, carryover, circuit concentration factors and coping with the often high suspended sediment loads in estuaries.

7.2.5 CW circuit

As ichthyoplankton entrainment represents one of the most difficult impacts to mitigate, attention must focus on aspects that are amenable to control:

- Initial site selection and intake positioning to minimise concentrations of ichthyoplankton in the abstraction zone.
- Design and operational aspects of the cooling system that influence survivorship of cooling system passage.

The first of these was dealt with earlier in this section. With regard to cooling system design, it is too early to make recommendations on pressure profiles within systems, as data are relatively sparse; at first sight, this would seem an unlikely route, as these conditions are largely determined by physical dimensions of the circuit which can probably not be changed much. Equally, temperature and exposure time probably cannot be altered. This leaves biocide toxicity as the main factor that might be altered. In some locations, where biofouling risk is low, it may be possible to avoid biocide application altogether (as is presently the case at Hinkley Point). Where biocide is used, it may be possible to modify dosing regimes to minimise entrainment mortality. This would require good knowledge of dose-response curves under simulated passage conditions (see Section 6.1.4). Where chlorine or other biocides are used, they are normally applied only for the active growth season, when water temperature exceeds 10^{o}C (roughly May-October), or as shown to be required by monitoring biological growth.

7.2.6 Outfall design and position

Optimising outfall location requires first the identification and mapping of sensitive habitats and biotopes, along with sensitivity criteria (such as maximum allowable exposure temperature or maximum allowable daily temperature change). Sources of data are identified in Section 6.3. Inspection of sensitivity maps will usually suggest a suitable location, which can then be investigated further using one of the plume or long-term heat field models listed in Section 6.3.4. Such analyses also seek to minimise recirculation risk between intake and outfall and may require several iterations.

The behaviour of the plume and areas swept by temperatures above +3°C ΔT (current maximum temperature uplift outside the plume in UKTAG draft standards) should be investigated by modelling; no sensitive habitat or biotope should be exposed to the plume for sufficiently long, or often or at temperature/ concentration sufficient to cause harm. Plume attachment to the shore should be avoided. The draft standards do not specify the allowable spread of the plume across an estuarine channel, whereas SEPA (2006) guidance states that the mixing zone for any type of discharge into transitional water should not take up more than half of the narrowest dimension. A more widely used criterion in overseas regulation that was also proposed by the authors of the technical report supporting the development of UKTAG draft standards (Turnpenny and Liney, 2006) requires the plume to occupy no more than 25 per cent of the channel cross-section for no more than five per cent of the time (allowing for some spread at slack water). It was excluded from the draft standards owing to potential enforcement difficulties, but this criterion provides clear guidance when interpreting plume model outputs and its adoption for this purpose is recommended.

Achieving an acceptable thermal field may require not only testing of alternative outfall positions but introducing variations in outfall design, for example by using diffusers.

7.2.7 Effects of long-term processes on cooling system viability

Climate change

This report does not cover climate change predictions or impacts but some indication is needed, in general terms, on what rising temperatures might mean for the operation of power plant cooling systems. Should background water temperature become one degree warmer than it is today then, by and large, water coming out of a direct-cooled station also will be one degree warmer. This is what currently happens but with a warming of 15ºC or so winter to summer, and the cooling system is designed to operate across this broad range of environmental conditions. As outlined in Chapter 2, maximum thermal efficiencies for steam turbines are achieved with low backpressure, typically 20 to 50 mbarg (2 to 5 kPa) equivalent to saturation values (condensation temperatures) of 18 to 33ºC. Efficient and economic transfer of heat from the tube surface to the cooling water requires a temperature differential ΔT of about 10ºC, so the cooling water source should not exceed 23ºC if 50 mbarg is to be maintained.

Table 7-3 Turbine backpressure and steam condensation temperature (saturation value) for a range of nominal CW intake temperatures (10ºC ΔT)

Backpressure mbar gauge	Condensation temperature ºC	Nominal intake or recool temperature ºC
20	17.5	8
30	24.1	14
40	29.0	19
50	32.9	23
60	36.2	26
70	39.0	29
80	41.5	32
84 Damhead Creek (design)	43.8	Air-cooled
150 Damhead Creek (extreme)	52.0	
186 Matimba S. Africa	59.0	

Direct-cooled CW systems can be designed for worst-case operation, like the highest recorded local sea temperature plus the maximum estimated re-circulation, plus a bit extra. Accommodating these marginal temperatures incurs additional cost and could result in excess cooling capacity for most of the station's life, particularly in winter when even conservatively designed stations can shut down one CW pump, thereby decreasing the rate of flow through the condensers with an increase in ΔT, assuming that the discharge consent allows this. Thus, a station can be designed in one of two ways: a low rate of abstraction and high ΔT, or a high rate of abstraction and low ΔT. Each approach has environmental advantages and disadvantages. It is usual, particularly today, to design around a reference temperature at the CW intake that will give adequate cooling for a percentage of the year whilst accepting some loss of efficiency (higher back pressure) for a few days, or even weeks, each summer. Electricity demand and prices are lower in summer anyway - although this could change with more air-conditioners being installed.

Current warming predictions appear to be around 0.1 to 0.2ºC per decade so, given a station lifetime of around six decades, it is unlikely that major changes to current design philosophy will be required. Towards the end of its working life, when predicted temperature rises might reach 1.0 ºC, the station might no longer be required to reach maximum output - assuming that the owner also operates newer stations. Efficiency per se has not been an issue with nuclear plant as it is for conventional plant since fuel represents a far lower proportion of generation costs. Nuclear plant is relatively inflexible in operation and historically has provided the baseload. If nuclear comes to represent a large proportion of UK installed capacity or if there is a major increase in intermittent renewables, such as wind, with privileged access to the grid, older nuclear plants could be forced into intermittent operation anyway.

The warm water discharge does not necessarily take longer to cool in summer, nor to spread more widely. Summer air temperatures are higher, as is the water/air saturation value, so evaporative cooling is enhanced. Consequently the atmospheric exchange coefficient AEC (net loss of heat from water to air) rises. On the other hand wind speeds tend to be lower in summer and this reduces cooling; some climate change models predict that these will increase.

There is little scope for amelioration. Station designers look for the coldest, usually the deepest, water source but the length of intake tunnel that can be built is constrained by economics. Recirculation of discharged water at some states of tide, or a slight build-up of heat in the area, might also be too expensive to avoid completely. The size of condensers, and hence abstraction rates, could be increased, but this also has monetary and environmental costs that may not be sustainable.

For river (tower-cooled) stations higher air temperature will enhance evaporative cooling, but at the cost of increased evaporative water loss and increased make-up requirement. The air/water ratio in the towers could be increased to obtain better recool but this would mean larger towers and more evaporative loss. High air temperature reduces the efficiency of dry towers and air-cooled condensers.

One of the many predicted effects of climate change is a modification to the seasonal pattern of rainfall and increased risk of water shortages in southern and eastern Britain. This could be a further reason to avoid wet tower-cooled freshwater sites.

Water levels can be expected to change as a result of climate change. Relative sea level rise is expected to be between 0.17 and 0.77 m by 2080 (UKCIP, 2006). Effects on cooling systems per se should be minor, or beneficial where dredged intake channels are used, in terms of increasing available water depth.

Geomorphology

Within the long life-cycle of a nuclear station, coastal processes in some dynamic locales may lead to major shifts in bathymetry that could threaten CW supply and alter patterns of heat dispersion. Moderate sedimentation can be dealt with by dredging but movement of offshore banks to cut off flow around the intake and outfall zones could reduce coastal water exchange and limit cooling capacity. It is therefore important to consider coastal processes at the planning stage for large plants.

7.3 Status of BAT definitions for cooling water systems

The European Commission Reference Document on Application of Best Available Techniques to Industrial Cooling Systems (BREF Cooling, December 2001) identifies direct cooling as BAT for large power plant cooling systems, stated as follows:

"*In an integrated approach to cooling an industrial process, both the direct and indirect use of energy are taken into account. In terms of the overall energy efficiency of an installation, the use of a once-through system is BAT, in particular for processes requiring large cooling capacities (e.g. > 10 MWth$_h$). In the case of rivers and/or estuaries once-through can be acceptable if also:*

- *extension of heat plume in the surface water leaves passage for fish migration;*
- *cooling water intake is designed aiming at reduced fish entrainment;*
- *heat load does not interfere with other users of receiving surface water.*

For power stations, if once-through is not possible, natural draught wet cooling towers are more energy-efficient than other cooling configurations, but application can be restricted because of the visual impact of their overall height."

It may be concluded from the BREF that direct cooling would not be BAT for large power stations if any of the three conditions were not met.

With regard to reduction of "entrainment" (for this read "entrainment and impingement" or "entrapment"), BAT requires positioning of the intake to avoid sensitive biotopes, spawning and nursery areas and fish migration routes. The BREF goes on to say "*From the applied or tested fish protection or repulsive technologies, no particular techniques can yet be identified as BAT*."

The validity of direct cooling as BAT has been challenged by consultants acting on behalf of Countryside Council for Wales (Cambrensis, 2008). Their report concludes: *"that both the developer and EAW[36] should fully consider alternatives to direct cooling for this installation at this site, particularly give significant developments in BAT since the European BREF guidance was last assessed in 2001, particularly in the USA, where they no longer regard direct cooling as best available technologies for coastal power stations".*

The Cambrensis conclusion has implications in a broader UK context and needs to be considered for future cooling requirements of large-scale power developments. Present indications are that direct cooling would be the method of choice at most, if not all, new UK nuclear developments.

The first point to consider is the relative thermal efficiency of available cooling methods, which has implications for the cost-per MWh of running the plant and levels of carbon emissions, assuming that any nuclear-generated electricity would displace the equivalent fossil-fuelled generation. The relative efficiencies remain as stated in Section 7.1.5 above, with direct cooling being the most efficient.

Secondly, if direct cooling is BAT on energy efficiency grounds, it needs also to meet the requirements of BREF listed above. The first of these deals with fish migration and would be met using advice identified in Section 7.2.6 above, which takes account of the latest understanding of temperature effects on fish migration developed as part of new WFD water temperature standards (UKTAG, 2008); this is augmented by new data on effects of temperature steps on juvenile fish migration, presented in Section 6.3.7.

Regarding impingement and entrainment of biota, the Cambrensis report notes advances in cooling technology since 1997 (the latest reference cited in BREF) but does not acknowledge progress made in entrapment mitigation. By 1997, UK power stations were operating only primitive fish return systems and deflection techniques for large water intakes had not advanced beyond the use of bubble curtains (Heysham I and II). Development of acoustic fish deterrent technology was in its infancy. In 2005, the Environment Agency and conservation bodies published their best practice guide on intake and outfall fish screening (Turnpenny and O'Keeffe, 2005), which identifies a range of techniques and combinations of AFD and FRR suitable for directly cooled coastal and estuarine plant. Now, a number of coastal power stations, along with numerous other water abstractions, use these techniques and the technology can be considered mature. Thus, an updated BREF might be expected to draw more positive conclusions on 'fish protection and repulsive technologies'.

The Cambrensis conclusion also refers to the situation in the USA, where there is a presumption against direct cooling (though not a ban) for large power plants. A distinct difference in the US approach has been the assumption of 100 per cent mortality of any fish eggs, larvae or juveniles entrained in plant cooling systems and discharged back to sea. UK studies have shown that substantial proportions survive CW system passage, potentially reducing the magnitude of entrainment impacts.

We therefore conclude that direct cooling may be the best environmental option for large power stations sited on the coast or estuaries, subject to current best planning, design and operational practice and mitigation methods being put in place, and meeting conservation objectives of the site in question.

[36] Environment Agency wales

7.4 Dealing with residual impacts

For a major project to proceed, impacts remaining after all mitigations have been applied must be deemed acceptable or compensated in some form. Recent years have seen much progress in the development of ecological compensation measures, both overseas and in the UK. The main requirements are that compensation should as far as possible be like-for-like and commensurate with, or an improvement upon, the level of impact. In practice this means exceeding the estimated loss, as replacement measures are often of lower quality than the original or may take time to develop.

Identifying suitable compensations sometimes requires imagination. Some years ago, a compensation plan was developed for a directly cooled plant proposal on the Usk Estuary. The environmental statement identified potential losses of salmon smolts to impingement and issues relating to upstream migration past the thermal plume. Compensation measures considered included a buy-out of the Usk salmon net fishery, improvements to fish passes and improvement of riparian habitat on the inland R. Usk. More recently, compensation habitat for UK port developments has been provided through managed coastal realignment projects.

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Glossary & list of abbreviations

ACC: Air-cooled condenser: a matrix of air-cooled finned tubes in which exhaust steam from the turbine is condensed directly.

AFD: Acoustic fish deterrent: propagation of underwater sounds to deflect fish from water intakes.

AGR: Advanced gas-cooled reactor: UK second generation carbon dioxide cooled, graphite moderated reactor with steam conditions comparable with conventional plant (500ºC/150 bar).

Approach velocity: Water velocity just upstream of a screen or water intake.

Band screen: Type of rotating fine filter screen, usually of 3-10 mm mesh, installed upstream of cooling water pumps and condensers to exclude marine detritus. Mesh is formed as 'conveyor' belt that rotates and is continuously backwashed to keep it clean.

BAT: Best available technology, as required under European Integrated Pollution Prevention and Control (IPPC) regulations.

Bottom resistance: Resistance of the sea bed or bottom of a river channel to flows.

BTA: Best technology available: US equivalent of BAT.

BWR: Boiling water reactor: A lightwater reactor in which the coolant/moderator is allowed to boil within the pressure vessel, providing low-quality steam at about 280ºC/65 bar.

CW: Cooling water/circulating water. Latter used mainly with tower cooling.

Drum screen: Type of rotating fine filter screen, usually of 3-10 mm mesh, installed upstream of cooling water pumps and condensers to exclude marine detritus. Mesh is formed as drum that rotates and is continuously backwashed to keep it clean.

Dry bulb temperature: Air temperaturea as measured by a thermometer freely exposed to the air but shielded from moisture and radiation.

EAV: Equivalent adult value: accounting method used in fish population dynamics, whereby the population value of a fish egg or juvenile is represented in terms of its probability of reaching adulthood.

Eddy viscosity coefficient: Vertical eddy viscosity coefficient relating average shear stress within a turbulent flow of water or air to the vertical gradient of velocity. It depends on fluid density and distance from river bed or ground surface. A horizontal eddy viscosity coefficient describes the horizontal mixing of mass and momentum.

EMU: Entrainment mimic unit: laboratory ecotoxicity test system that simulates conditions experienced by entrained biota during cooling system passage.

Entrainment: Passage of entrapped organisms that penetrate CW screens (typically zooplankton including ichthyoplankton and phytoplankton), via pumps, heat exchangers and other components of the CW circuit and back to the receiving water.

Entrapment: Inadvertent entry into the CW system of aquatic organisms caused by the ingress of water.

Fish bucket: Modified band or drum screen elevator within an FRR system.

FRR: Fish recovery and return: system using band or drum screens modified for safe fish handling, including their return to the source water body.

Ichthyoplankton: Early life-stages of fish contained within marine plankton.

Impingement: Retention of entrapped organisms on CW intake screens employed to prevent debris entering the CW heat exchangers.

Latent heat: Amount of energy released or absorbed by a chemical substance during a phase transition (e.g. from liquid to gas phase).

Launder: Troughs, channels or pipes used to carry trash that has been backwashed from the fine screens.

Magnox: First generation UK carbon dioxide cooled, graphite moderated reactor. Named for the magnesium alloy fuel can **Mag**nesium no**n-ox**idisable.

Make-up: Water used to replace that lost to evaporation in wet-cooling towers.

Manning's roughness coefficient: Empirically derived roughness coefficient representing resistance to flows in open channels and flood plains. Its value depends on many factors, including river-bottom roughness, stage of flows, flow velocity and vegetation.

MCW: Main cooling water system.

MWe: Megawatts (electric): rated capacity of a turbo-alternator or installed capacity of an entire power station.

MWth: Megawatts (thermal): rated heat output of a boiler, reactor or entire power station. About two or three times higher than the MWe.

Purge: Water used to remove dissolved and suspended solids that have become concentrated in tower cooling systems.

PWR: Pressurised water reactor.

Wet bulb temperature: The lowest temperature a wetted body will attain when exposed to an air current.

Appendix A: CW intakes at nuclear and other large (over 1,000 MWe) UK power stations

Name	Fuel or Type	Current Status	Installed Capacity Mwe	Maximum CW flow m^3s^{-1}	Cooling Water Source	Intake Position	Moving Screens	Fish Protection or Return
Aberthaw	Coal	Operating	1,500	67	Sea	Offshore	Pressure	No
Barking	CCGT	Operating	1,000	Towers	FW River	Inshore		R
Berkeley	Magnox	Closed	332	26	Estuary	Inshore		P & R
Blyth A&B	Coal	Closed	2,140	65	Estuary	Onshore	Band	No
Blyth C	Coal	Proposed	1,600	65*	Estuary	Onshore	Drum?	
Bradwell	Magnox	Closed	312	26	Estuary	Offshore	Disc	No
Cockenzie	Coal	Operating	1,200	38	Sea	Onshore		No
Connahs Quay	CCGT	Operating	1,380	Towers	Estuary	Inshore	W - wire	No
Cottam	Coal	Operating	2,008	Towers	FW River	Onshore		
Didcot A	Coal/gas	Operating	1,925	Towers	FW River	Onshore	Drum	
Didcot	CCGT	Operating	1,370	Towers	FW River	Onshore	Drum	
Dounreay	PFR, DFR	Closed	variable		Sea	Onshore		No
Drax	Coal/ biomass	Operating	3,870	Towers	FW River	Onshore		No
Dungeness A	Magnox	Closed	560	27	Sea	Offshore	Band	No
Dungeness B	AGR	Operating	1,320	40	Sea	Offshore	Drum	No
Eggborough,	Coal	Operating	1,960	Towers	FW River	Onshore		
Fawley	Oil	Operating	2,000	60	Estuary	Onshore	Drum	P & R
Ferrybridge	Coal	Operating	1,955	Towers	FW River	Onshore		
Fiddlers Ferry	Coal	Operating	1,989	Towers	River	Onshore		
Grain	Oil	Operating	2,640	58	Estuary	Inshore	Pressure	No
Hartlepool	AGR	Operating	1,320	33	Estuary	Onshore	Drum	No
Heysham 1	AGR	Operating	1,320	33	Estuary	Onshore	Drum	No
Heysham 2	AGR	Operating	1,320	50	Estuary	Onshore	Drum	No
Hinkley Point A	Magnox	Closed	580	40	Estuary	Offshore	Band	No
Hinkley Point B	AGR	Operating	1,320	30	Estuary	Offshore	Drum	No
Hunterston A	Magnox	Closed	360		Sea	Inshore	Band	No
Hunterston B	AGR	Operating	1,320	28	Sea	Offshore	Drum	No
Inverkip	Oil	Closed	2,028	45	Estuary	Onshore	Drum	No
Ironbridge	Coal	Operating	972	Towers	FW River	Onshore		
Kingsnorth	Coal/oil	Operating	2,000	64	Estuary	Onshore	Drum	No
Kingsnorth	Coal	Proposed	1,600	64*	Estuary	Onshore	Drum?	Yes
Littlebrook	Oil	Operating	1,980	58	Estuary	Inshore	Drum	No
Longannet	Coal	Operating	2,640	90	Estuary	Onshore	Drum	No

Name	Fuel or Type	Current Status	Installed Capacity Mwe	Maximum CW flow m^3s^{-1}	Cooling Water Source	Intake Position	Moving Screens	Fish Protection or Return
Oldbury	Magnox	Closed	626	27	Estuary	Onshore	Drum	R
Pembroke	Oil	Closed	2,000	60	Estuary	Onshore	Drum	R
Peterhead	Oil/gas	Operating	1,320	28	Sea	Onshore	Drum	No
Ratcliffe	Coal	Operating	2,000	Towers	FW River	Onshore		
Rugely	Coal	Operating	1,006	Towers	FW River	Onshore		
Saltend	CCGT	Operating	CHP	Towers	Estuary	Onshore		
Sizewell A	Magnox	Closed	650	34	Sea	Offshore	Drum	No
Sizewell B	PWR	Operating	1,320	48	Sea	Offshore	Drum	R
S.Humber Bank	CCGT	Operating	1,300		Estuary	Offshore		
Teesside	CCGT	Operating	CHP	Towers	Estuary	Onshore		
Tilbury	Coal/oil	Operating	1,400	53	Estuary	Inshore	Drum	R
Tilbury C	Coal	Proposed	1,600	53*	Estuary	Inshore	Drum?	
Torness	AGR	Operating	1,320	47	Sea	Onshore	Drum	No
Trawsfynydd	Magnox	Closed	560	40	FW Lake	Onshore		
West Burton	Coal	Operating	2,000	Towers	FW River	Onshore		
Wylfa	Magnox	Operating	1,000	67	Sea	Inshore	Drum	No

Italics denote proposed 2x 800 MWe coal-fired stations for which the CW demand will be "within the existing" consent.
Tower-cooled station often do not abstract continuously; overall their abstraction is about three per cent that of a comparable direct-cooled station. Of this, one per cent is "consumed" by evaporation and the other two per cent is returned to the waterway as purge (or blowdown) to limit the build-up of dissolved and suspended solids in the tower circuit.
*Installed, not current declared capacity, is shown since this better indicates potential cooling capacity of the site. Similarly it is assumed that all of the installed CW pumps are running.

Appendix B: Cooling methods, arranged in order of decreasing water demand

Description	Source	Description	Comments
Direct cooling		Discharged heat is dispersed and diluted in a large volume of water; heat ultimately lost to atmosphere. Intake and outfall sited to avoid or minimize early recirculation of still-warm water. Outfall can be arranged to maximise spread of warm, low-density effluent at surface or to maximise dilution into the water body.	
	Sea	Problems generally relate to hydraulic gradient across tidal range.	Widely used
	Estuary	Problems generally relate to hydraulic gradient across tidal range. Issues such as migratory fish and heat dispersion.	Widely used
	Tidal reservoir	Where length of intake channel or culvert would be excessive or the hydraulic gradient at low water would be too great.	Few examples; Oldbury UK
	FW river	Lack of suitable sites in UK. Issues of fish and of heat dispersion. Discharges of effluents e.g ash-handling, radionuclides, biocides into potable supply.	Widely used where large rivers available
	Lake	Issues of fish and of heat dispersion. Discharges of effluents e.g ash-handling, radionuclides, biocides into potable supply.	Widely used e.g. by all Canadian nuclear stations
	FW Reservoir	Extended or created for use for cooling – generally with bunds or lagoons to direct effluent along the longest path back to intake.	Trawsfynydd UK North Anna US
	Canals	A linear form of the above, with several broad, shallow parallel canals. Can be augmented by sprays.	Turkey Point Florida
Spray assisted direct cooling	Spray pond	Essentially a pond with fountains. Water droplets lose heat by contact with air and by evaporation. Fountains also limit formation of a blanket of saturated warm air over the pond that would hinder heat loss from its surface. Overall water volume still large in relation to generation.	Central Electricity Research Labs; proposed Bradwell HTR
Tower Cooling 1. Wet towers		Typically specified where water source could not sustain once-through cooling; however, it also allows more flexibility in siting (novel water sources are described below). Towers enhance air/water contact and minimise the volume of water in circulation. Effectiveness is related to *wet bulb* temperature and the air:water ratio. A high air:water ratio pemits a close approach to wet bulb temperature but needs large airflow and large towers.	

1a. Natural draught towers	Spray towers	Similar to a domestic shower. Water droplets fall freely inside an open-topped tower, entraining air that is carried down (co-current) and escapes through the louvered sides. Cooled water collects in the basin under the tower.	Small units only, less commonly used now.
	Filled towers	Use fill (pack) beneath the water distribution system to prolong air/water contact time. *Splash pack* consists of tiers of horizontal timber or plastic laths onto which water showers, splashing, running over the lath as a thin film and dripping onto the lath below. This has largely been replaced by film-forming pack (*film pack*) down which water flows as a thin film. This yields lower recool temperatures but is more prone to fouling than the essentially self-cleaning splash pack. Towers come in all sizes but the large concrete hyperbolic ones probably are the most familiar. The “works” (water distribution and pack) occupy only a fraction of the height of the tower – the rest acts as a chimney drawing cold air through the pack (counter-current). The water collects in a pond at the base of the tower.	Large hyperbolic towers of this type are often the most evident part of a power station. Widely used
1b. Mechanical draught towers		These are lower than natural draught towers since they use fans (forced or induced draught) to push/pull air through the (usually) film-pack: these fans can be noisy. The pitch and/or speed of the fans can be varied and air distribution, in some towers, can be adjusted by doors. One disadvantage of these low towers is that plume formation occurs close to the ground and can result in fog or icing on nearby roads.	Commonly used for CCGTs. Built in banks to provide necessary capacity
	Crossflow	Air moves horizontally through the pack. The tower is usually lower than the counterflow pattern.	
	Counterflow	Air moves vertically through the pack, against the flow of water. This gives more efficient heat transfer since the coolest water contacts the coolest air.	
2. Hybrid wet and dry towers		These are mainly based on mechanical draught towers and have two sections. The upper *dry* section contains tubes, usually of the same material as condenser tubes, through which the cooling water flows. Here, heat is lost by conduction to air – sometimes referred to as the *radiant* section. The lower section is a standard wet tower. In some cases the dry section serves merely to heat the moist air from the wet section, thereby minimising low-level plume formation. In some situations it is possible to rely upon the dry section alone in cold weather.	
3. Hybrid direct/indirect systems		Often called auxiliary tower systems. One or more towers are used to augment or completely replace the cooling capacity e.g. during periods of low river flow or when river temperature exceeds some preset value.	Barking UK

4. Dry towers		These are essentially large car radiators, with the water to be cooled circulating in finned (to increase the area of cooling surface) tubes. The radiator panels may be sited around the base of a natural draught hyperbolic tower or rely upon mechanical draught.	
	Condenser coolant cooling	The circuit is as for a wet tower, except that the recirculating coolant does not come into direct contact with air. Thus there is no evaporative loss, but by the same token, no evaporative (latent) cooling either. (Indirect dry tower)	Kendal SA (Eskom)
	Jet (spray) condenser	The condenser has no tubes and no separate cooling water. The steam exiting the turbine is condensed by contact with a spray of cold water. Most of the condensate is recycled to the boiler but a proportion is pumped through finned coolers in a dry tower and provides the condenser spray.	Rugely (UK) used radiators at the base of a hyperbolic tower
	Helper tower	A dry tower connected in series or in parallel with a wet tower to improve its thermal performance.	Proposed Callide B (Australia)
5. Wet surface air coolers WSACs		Radiators rely upon a large airflow to cool (remove sensible heat) from the fluid in the tubes. Spraying water onto the tube surface adds evaporative cooling and enables the fluid recool temperature to approach wet bulb (wind-chill factor) rather than dry bulb (air) temperature. The spray water can be of extremely low quality. Wet-surface technology can be applied to a range of dry coolers, often temporarily to overcome extreme weather conditions.	
5. Air-cooled condensers		Most of the systems described above rely on two-stage cooling, using cooling water as an intermediate fluid to carry heat away from the condenser. Air-cooled condensers accept steam directly from the back-end of the turbine and condense it in large arrays of finned tubes cooled by large fans. Initially these were seen as a solution for electricity generation in arid areas but increasingly are being advocated as a solution to problems over abstraction and discharge of cooling water.	Mystic Harbour, Boston USA (on water's edge). Damhead Creek, Medway UK
6. Novel sources of cooling water		Not all are necessarily novel. These are sources that are generally unfit for potable or agricultural use. Their burden of dissolved solids can pose problems in evaporative systems but can be managed by side-stream treatment or purge back to source.	
	Treated sewage effluent	Tertiary-treated sewage effluent was used in some UK tower circuits (e.g Kingston) since at least the 1950s. There is good synergy since areas of population needing electricity also produce effluent.	

	Saline aquifers	Underlie large areas of arid USA.
	Mine water	Plentiful in areas such as Kentucky.
	Process water	Ranges from run-off from agricultural land to effluents from food-processing, manufacturing and water entrained by oil and gas extraction.
7. Water recovery		Initially developed to provide feedwater for steam-electric generation in arid areas.
	Flue gas	Removal of water from combustion gases using recycled liquid desiccant (see carbon dioxide capture).
	Coal drying	Softer coals – and biomass - contain moisture that reduces gross efficiency (since water vapour with latent heat goes up the chimney). Pre-drying, using low-level waste heat, reclaims this water and improves efficiency.
	Diffusion tower	One per cent of water circulating in a wet cooling tower is lost by evaporation. No matter how poor the circulating water quality, this loss is essentially distilled water that can be recovered and polished for boiler feedwater or simply returned as make up to the tower.

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CASE STUDY

SPA

Fish Deflection for Thermal Power Station

Doel Nuclear Power Plant, Belgium

Figure 1

Water Intakes and Outfall at Doel Nuclear Power Plant, Belgium

Doel Nuclear Power Station is located on the River Scheldt, close to Antwerp in Belgium and is operated by Electrabel. The station comprises four pressurised water reactors (PWR); Doel 1&2, which each produce 433 MW, were commissioned in 1975; Doel 3, which produces 1006 MW was commissioned in 1982, while Doel 4 produces 1039 MW, and was commissioned in 1985.

Doel 1&2 share a common intake, which is connected to the shore by a concrete jetty, while the intake for Doel 3&4 is located approximately 250m offshore, with an underground tunnel connecting to the onshore screens. The intakes can be seen in the photo below, separated from the outfall by the breakwater.

Electrabel approached Fish Guidance Systems Ltd in 1996 to help reduce the numbers of fish that were being drawn into Doel 3&4 intake, as the entrained fish were creating both environmental and operational issues for the station. The main species being affected were herring (*Clupea harengus*) and sprat (*Sprattus sprattus*), both members of the clupeid family and so expected to respond well to an acoustic fish deflection (AFD) system.

In 1997, following acoustic (PrISM) modelling, a SPA system was designed and installed on the offshore cooling water intake. The Sound Projectors were positioned so that they create a repellent sound field close to the water intake openings, so that the fish are deflected into the streamlines of the main river flow that passes the intake. The system comprised of –

- 1 off FGS Model 1-08 Signal Generator
- 18 off FGS Model 400 Amplifiers
- 2 off Weatherproof Control Equipment Enclosures
- 2 off Junction Boxes
- 20 off FGS MkII 30-600 Sound Projectors
- 20 off ‘Flying Leads’

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Figure 2

Acoustic Fish Deflection System deployed on Doel 3&4 Cooling Water Intake

Figure 3

Detail of winch system used for deploying Sound Projectors

- 10 off spare MkII 30-600 Sound Projector Internal Units for maintenance

To allow servicing of the fish deterrent system whilst the station is still operating, deployment frames were installed to lower the Sound Projectors into their optimum position and to allow them to be raised for routine inspections and maintenance.

The acoustic installation has undergone a number of evaluation trials by researchers from Belgium's Leuven University. The independent trials have shown a reduction in the target species of up to 98%, with an overall reduction for all fish species of 80% -

Herring (*Clupea harengus*) - 98%

Sprat (*Sprattus sprattus*) - 97%

Bass (*Dicentrarchus labrax*) - 89%

Gobies (*Gobidae*) – 75%

Overall reduction - 80%

(Nov. 1999 trial)

The system was originally supplied on a lease-purchase based, with the maintenance of the system included as part of the lease cost. Since the lease period ended in 2007 FGS has continued to provide a service contract for the system, ensuring it is maintained on an annual basis.

For a copy of the independent trial results for this system, or further information on this site and other fish deflection systems please contact FGS.

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CASE STUDY

SILAS

Thermal Power Station

Pembroke, Wales

Pembroke B Combined Cycle Gas Turbine (CCGT) Power Station is the largest power station to be built in the United Kingdom since Drax Power Station was completed in 1986, and is the largest CCGT station in Europe. The station is located on Pennar Gut, a small bay feeding Milford Haven in West Wales and is built on the same site as the original oil-fired station that was closed in 1997.

The new power station is operated by RWE npower and produces 2000 MW from five 400 MW generator units, each using direct seawater cooling abstracted from the original cooling water intake. Milford Haven is surrounded by the Pembrokeshire National Park, and Milford Haven itself falls within the boundary of the Pembrokeshire Marine Special Area of Conservation (SAC). Of particular note in Pembrokeshire SAC are eel, salmonids, lampreys and grey seal, but all fish are considered important at the site since they contribute to the food chain within the SAC.

Even though the original cooling water intake has been used for the new station, it was not designed to modern environmental standards, and the increased emphasis on protecting the marine environment resulted in the need to minimise the impact of water abstraction on the SAC. RWE npower was therefore required to install the Best Available Technology (BAT) to screen the intake in accordance with the Environment Agency's Best Practice Guide for the Screening of Intakes. The most suitable system for large scale coastal abstractions is deemed to be an acoustic fish deterrent (AFD) system combined with a fish recovery and return (FRR). However, to maximise the effectiveness of the system against European eel (*Anguilla anguilla*), which are now protected under The Eels (England and Wales) Regulations 2009, the Environment Agency required that 'strobe' lights be incorporated into the deterrent system.

Fish Guidance Systems was therefore commissioned to supply a suitable fish

Figure 1

Pembroke Power Station with the intake in the foreground

Figure 2

Pembroke Power Station with the intake and Pennar Gut in the background

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Figure 3

FGS MkIII SILAS® Sound Projectors at Pembroke Power Station

deterrent system and FGS proposed a Sound Projector Array (SPA) system should be installed. However, due to the large size of the intake, and the need to install 'strobe' lights, the latest MkIII version of the SPA system was proposed, which uses FGS's patented SILAS® technology.

The MkIII Sound Projectors are based upon the previous MkII units, but each Sound Projector contains a digital signal, amplifier and other electronics required to operate and monitor the projector. In addition, the Sound Projectors incorporate FGS's High Intensity Lights, which are synchronised to flash with the acoustic signal. This has the benefit of not only deflecting eels, but also of increasing the overall effectiveness of the system for other fish species in the vicinity of the intake.

Since the original intake was to be used the fish deterrent system had to be retrofitted to the original structure, which comprised 17 inlet gates, and in order to maintain an intake velocity of less than 0.3 m/s, the whole of the intake is used to abstract the required water. The projectors could not be located in the main flow as they might impact the flow of water into the intake, and so they were mounted on the buttresses that separate the inlet gates. As a result, there are 18 columns of Sound Projectors, with one column on each buttress.

PrISM modelling was carried out to determine the optimum number of Sound Projectors for the system, which initially indicated that 54 Sound Projectors were required to screen the intake however, the Environment Agency also requested redundancy be incorporated into the system owing to the ecological sensitivity of the site. As a result four Sound Projectors were specified for each column, with a total of 72 Sound Projectors being installed on the intake.

In summary, the system installed at Pembroke comprises of –

- 1 off FGS SILAS® Control Unit
- 6 off FGS Model 3000 Power Supply Units
- 18 off Power and Communication Hubs
- 72 off FGS MkIII SILAS® 30-600 Sound Projectors with integrated High Intensity Light Rings
- 72 off 'Flying Leads'
- 2 off Communication cables
- 6 off Power Supply cables

Four drum screens and a fish recovery and return system were installed by EIMCO Water Technologies Ltd (now OVIVO UK Ltd) to complete the screening of the intake.

The design of the deployment system for the Sound Projectors was carried out by EIMCO,

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Figure 4

FGS SILAS® System installed at Pembroke Power Station

based upon designs used by FGS at previous installations. The projectors are bolted to sliders that are mounted upon vertical rails, and a lifting wire runs up to the surface so that the projectors can be raised for cleaning and maintenance. The projectors are raised by using the permanent overhead crane installed for maintenance of the main intake, and a walkway runs in front of the projectors, to enable access from the power station, rather than having to use a boat. In this way, the number of personnel required to access the projectors was reduced, as well as the time required to maintain the system.

The SILAS® Control Unit was set up to provide a 'Profibus™' link to the general monitoring system within the power station, so that the system could be monitored remotely from the main Control Room at all times.

Currently no trials have been carried out, due to the need to switch off the system as part of the evaluation, but discussions are on-going with Natural Resources Wales (formerly the Environment Agency) on how best to achieve this, and the results will be reported at a later date.

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Field evaluation of a sound system to reduce estuarine fish intake rates at a power plant cooling water inlet

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An acoustic deterrent system producing 20–600 Hz sound was used to repel estuarine fishes away from a power station cooling water inlet. During sound emission, total fish impingement decreased by 60%. The avoidance response varied among species from no effect to highly efficient deflection. *Lampetra fluviatilis* and Pleuronectiformes were less affected by the sound system while the deflection of clupeoid species was particularly effective. Average intake rates of *Clupea harengus* and *Sprattus sprattus* decreased by 94·7 and 87·9%, respectively. The results were explained as a function of species-specific differences in hearing ability and swimming performance. In general, species without swimbladders showed no or a moderate response while intake rates of species with accessory structures increasing the hearing abilities, such as a swimbladder or a functional connection between the swimbladder and the inner ear, were significantly reduced during test periods.

Key words: acoustic fish deterrent; *Clupea harengus*; cooling water intake; low frequency sound.

INTRODUCTION

The abstraction of cooling water by power plants causes a wide range of ecological impacts on aquatic communities. Effects are situated both within the cooling-water circuit and in the cooling water receiving water body. Thermal loading related to cooling water discharges directly interferes with physiological processes of the biota, such as enzyme activity, feeding, reproduction, respiration, growth and photosynthesis (Hadderingh *et al.*, 1983). Behavioural changes (attraction or avoidance) are commonly observed in organisms subjected to thermal discharges as well (Kennish, 1992). Of greater potential impact on the aquatic communities than waste heat discharges, however, are the losses of various life-history stages of invertebrates and fishes due to impingement on intake screens or entrainment through cooling systems. It is not uncommon for millions of fishes and crustaceans

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to become impinged on power plant intake screens each year (Kennish, 1992) and many more eggs, plankton and larvae of invertebrates and fishes are killed. The absolute numbers are $>10^{10}$ individuals annually at some power plants (Kennish, 1992). In addition to impingement and entrainment, adverse effects on aquatic organisms may arise from the use of biocides to control fouling organisms.

To reduce the impact of cooling water abstraction on fish populations in the surrounding waters, different methods and devices have been introduced with variable success. The major deflection methods at power stations include visual stimuli (*e.g.* air-bubble screens, lights or strobe lights), water velocity and pressure changes, electrical shocks and sounds. Mechanical exclusion devices use fine-screens surrounding an intake from which cooling water is drawn.

The Doel nuclear power plant (Belgium) withdraws cooling water from the Scheldt Estuary, an important nursery for young-of-the-year marine and freshwater fish species (Maes *et al.*, 1998). The density of fishes in the cooling water circuit averaged 64 fishes per $1000\,m^3$ in the period between 1991 and 2001, resulting in the loss of *c.* 50 million fishes each year. In addition, a similar number of crabs and shrimps are retained on the filter screens. To reduce the impact of the power plant cooling water inlet on the estuarine fish populations, it was proposed to install and evaluate a fish deflection system based on sounds to guide or deter the fishes away from the cooling water inlet. This proposal was based on two major observations. First, it was assumed that conventional deflection devices based on light stimuli or electrical fields, if installed, were not likely to be efficient due to the high natural turbidity and the salinity of the water. Also mechanical exclusion devices to prevent fishes from entering the intake were considered impractical. Second, trials on site showed that representatives of the Clupeidae (herrings), Osmeridae (smelts) and Cyprinidae (carps) were moderately to extremely sensitive to mechanical contacts and suffered post-intake mortalities that were higher than those for other fish species. The aim was to deter these highly vulnerable fish species.

The objective of this study was to test the efficiency of an acoustic fish deterrent (AFD) system placed at the cooling water intake to avoid fishes entering. Previous studies have used sound in an attempt to influence the movement of fishes near to power plant cooling water intakes or at hydropower facility turbines. These studies, however, have concentrated mainly on American salmonids (Knudsen *et al.*, 1997) and clupeids (Haymes & Patrick, 1986; Ross & Dunning, 1993) and ranged from totally unsuccessful in controlling behaviour, to demonstrating potential usefulness for a few species under certain conditions (Anon., 1995). Few other fish groups have been tested in a systematic way to determine if they would avoid low frequency sound (Anon., 1995). Knudsen *et al.* (1997), however, recognized the need for large-scale field studies to test the true effectiveness of infrasound screens under natural conditions. Accordingly, this paper is one of the few reporting on the avoidance response of an entire fish community to the production of underwater infrasound.

MATERIALS AND METHODS

The Doel nuclear power plant is located on the west bank of the brackish water part of the Scheldt Estuary (51°15′ N; 04°17′ E) (Maes *et al.*, 1998). Its cooling water intake is situated 2 m above the bottom and withdraws $25{\cdot}1\ m^3 s^{-1}$ of water through five intake apertures measuring $4{\cdot}0 \times 2{\cdot}4\ m^2$ each. The intake is located in open water at 50 m off the shore during low tide and 200 m at high tide. Water is transported through a 540 m long pipeline to a communicating reservoir at the site. Here, fishes, crustaceans, plant material and debris are removed from the cooling water by vertical travelling water screens with a square mesh-size of 4 mm and afterwards flushed into a container, the sampling point. The residence time of fishes within the system *i.e*. the time animals spend between the intake point in the river and the sampling point is at least 20 min. Goldfish *Carassius auratus* (L.), measuring between 7 and 23 cm in total length (L_T) were used to estimate more exactly the residence time of fishes by releasing 246 live fish into the cooling water inlet. After 20 min, 69% of the fish were recaptured, after 1 h, 80%.

In 1997, a sound projector array AFD system was designed by Fish Guidance Systems Ltd. (FGS, Southampton, U.K.) and installed near the offshore intake. In total, 20 large FGS Mk II 30–600 sound projectors (600 W each) were installed to create a repellent sound field close to the water intake openings to cause passing fishes to 'veer away'. A multiple signal generator, programmed with eight different sound signals, was used to avoid resident species habituating to any one sound signal. The sounds comprised frequencies within a range of 20–600 Hz repeated every 0·2 s. These were emitted from the sound projectors *via* a flexible neoprene membrane, producing a nominal sound pressure output of 174 dB (reference pressure: 1 μPa). The membrane is protected from external damage by a conical grill of concentric rings. Depth compensation is provided by an air reservoir. A preliminary evaluation in 1997–1998 yielded no significant reduction in numbers of fishes entering the intake and the sound projectors were relocated and installed at the intake. After reinstallation, 15 trials were held between October 1998 and October 2001 to evaluate the efficiency of the AFD system (Table I). Each trial lasted 48 h. The AFD system was operating during the first day (24 h) of the trials and was turned off during the second day (24 h). Fishes and crustaceans were collected in nets, identified and counted. Gobies of the genus *Pomatoschistus* were not identified to species level.

A *t*-test for dependent (or paired) samples was performed on the $\log_{10}$-transformed data to examine the difference in the total catch in numbers observed between control days without sound projection and days with sound projection. Differences were significant at $P < 0{\cdot}05$. Fifty individuals, caught over the entire test period, were used as the minimum sample size for statistical testing. The reduction R in intake numbers was

TABLE I. Sampling dates of the trials and tide-averaged temperature and salinity

Sample number	AFD on	AFD off	Temperature (° C)	Salinity
1	26 October 1998	27 October 1998	14·8	7·5
2	9 November 1998	10 November 1998	10·2	1·0
3	23 November 1998	24 November 1998	7·8	1·4
4	7 December 1998	8 December 1998	5·4	1·7
5	15 December 1998	16 December 1998	6·7	1·8
6	12 January 1999	13 January 1999	7·9	1·9
7	2 November 1999	3 November 1999	12·6	11·6
8	1 December 1999	2 December 1999	8·8	9·4
9	26 January 2000	27 January 2000	6·6	1·5
10	10 June 2000	11 June 2000	18·7	9·2
11	29 November 2000	20 November 2000	13·0	5·2
12	19 December 2000	20 December 2000	11·2	2·0
13	28 February 2001	1 March 2001	10·1	0·9
14	26 September 2001	27 September 2001	19·0	7·0
15	23 October 2001	24 October 2001	17·2	3·5

AFD, acoustic fish deterrent.

expressed as a percentage: $R = 100 - 100\ n_{on}\ n_{off}^{-1}$, where n_{off} is the total catch of the control samples (sound off) and n_{on} is the total catch of the test samples (sound on).

Additionally, the influence of salinity and temperature on the species-specific efficiency of the AFD system was tested. The contribution of each of these variables on the variation in percentage reduction was assessed using multiple linear regression models. On selected occasions, the L_T (cm) of herring *Clupea harengus* L. and sprat *Sprattus sprattus* (L.) were measured to examine differences in average fish size during test and control periods. The hypothesis of no difference between sound-on and sound-off conditions was tested using one-way ANOVA.

RESULTS

During the survey *c*. 350 thousand fishes were sampled comprising 24 families and 41 species. Gobies were not identified to species level but three species occurred in the screen samples: the marine gobies *Pomatoschistus minutus* (Pallas) and *Pomatoschistus lozanoi* (de Buen) and the estuarine resident *Pomatoschistus microps* (Krøyer). Marine species, both marine estuarine opportunistic species and marine vagrants, dominated the fish fauna in terms of diversity and abundance. Other life cycle categories including freshwater fishes and diadromous species were less abundant.

Total fish impingement decreased by 59·6% during the AFD operation. The overall reduction of fish impingement was mainly due to a reduction in the number of gobies since these species represented 78% of the entire catch. Most species that were available for entrapment by the cooling water intake during the study period, however, decreased their intake numbers. The difference in intake numbers between sound-on and sound-off conditions was significant for nine species or taxa (Table II): herring, sprat, white bream *Abramis bjoerkna* (L.), smelt *Osmerus eperlanus* (L.), bass *Dicentrarchus labrax* (L.), perch *Perca fluviatilis* L., sole *Solea solea* (L.), flounder *Platichthys flesus* (L.) and gobies of the genus *Pomatoschistus* sp. In particular, the deflection of clupeids was successful. For herring, AFD operation resulted in an average catch reduction of 94·7%. In periods of maximum herring abundance in the estuary, >99% were deterred (Fig. 1). Reduction in the number of sprat during test periods averaged 87·9%. White bream was the only cyprinid for which numbers drawn in were significantly reduced but numbers of other representatives of this family were rather low. In Percidae, AFD operation yielded a significant reduction in the catch of perch by 51·2%. Numbers of the pikeperch *Stizostedion lucioperca* (L.) decreased by almost 70% but the reduction was not consistent over the different trials and hence, not significant (Table II). Results for flatfishes varied, with a significant reduction in sole and flounder. The catch of dab *Limanda limanda* (L.) did not vary between the test and control conditions. Pipefishes, sticklebacks and mullets did not show any avoidance reaction to the AFD system (Table II). For all other species including gadoids, numbers were fairly low and no (statistical) conclusions were drawn. Data for these species are therefore not presented.

Multiple regression of temperature and salinity as independent variables and the percentage reduction of the nine species that responded significantly to sound emission as dependent variable did not yield significant models (multiple regression analysis; $n = 15$; $P > 0·05$). Temperature and salinity modify the speed

TABLE II. Percentage reduction in the total catch of fishes impinged by the cooling water intake of the Doel nuclear power plant during the experiments. The total catch (sum) and the daily mean ± S.E. sample size and ($n = 15$) are given for test periods (AFD on) and for control periods (AFD off). Fish species are subdivided into a group with significant reductions during sound projection (*t*-test for dependent samples; $n = 15$; $P < 0.05$) and a group with statistically insignificant reductions (*t*-test for dependent samples; $n = 15$; $P > 0.05$). Data for species of which <50 individuals were caught during the trials are not given

Species	%	AFD on		AFD off	
Significant reductions		**Sum**	**Mean ± S.E.**	**Sum**	**Mean ± S.E.**
Clupea harengus	94·7	2095	139·7 ± 49·2	39 893	2659·5 ± 748·7
Sprattus sprattus	87·9	4298	286·5 ± 83·1	35 647	2376·5 ± 672·6
Dicentrarchus labrax	75·6	447	29·8 ± 7·4	1831	122·1 ± 26·4
Osmerus eperlanus	53·5	260	17·3 ± 12·4	559	37·3 ± 25·6
Perca fluviatilis	51·2	168	11·2 ± 4·8	344	22·9 ± 9·3
Solea solea	46·6	86	5·7 ± 4·7	161	10·7 ± 7·6
Pomatoschistus sp.	46·1	91 652	6110·1 ± 2715·4	169 938	11 329·2 ± 4694·1
Abramis bjoerkna	40·1	82	5·5 ± 4·0	137	9·1 ± 3·7
Platichthys flesus	37·7	129	8·6 ± 6·0	207	13·8 ± 9·3
Insignificant reductions					
Liza ramada	75·8	109	7·3 ± 2·2	450	30·0 ± 17·1
Stizostedion lucioperca	69·8	74	4·9 ± 2·0	245	16·3 ± 9·3
Syngnathus rostellatus	43·3	394	26·3 ± 17·4	695	46·3 ± 45·6
Anguilla anguilla	37·3	141	9·4 ± 3·0	225	15·0 ± 5·1
Pungitius pungitius	7·1	195	13·0 ± 4·2	210	14·0 ± 4·7
Lampetra fluviatilis	5·9	32	2·1 ± 1·4	34	2·3 ± 1·2
Gasterosteus aculeatus	1·4	1542	102·8 ± 32·1	1564	104·3 ± 32·4
Limanda limanda	0·0	111	7·4 ± 7·4	111	7·4 ± 6·5

AFD, acoustic fish deterrent.

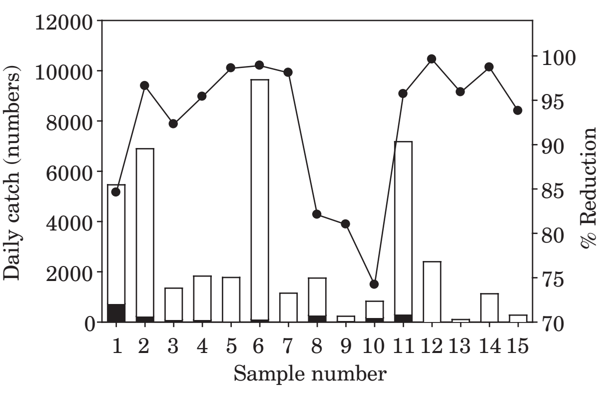

FIG. 1. Numbers and reduction (●) in the daily catch of herring during sound emission (AFD on, ■) and during controls (AFD off, □). Dates corresponding to the sample numbers are given in Table I.

TABLE III. Difference in total length in herring and sprat during sound emission (AFD on) and during the control period (AFD off). Differences in L_T were tested for statistical significance using one-way ANOVA on two groups. Values for the number of observations (n) and the probability level are also given

Species Date	AFD on L_T (cm)	AFD off L_T (cm)	n	P
Clupea harengus				
2 Feb1999	8·2	8·6	269	<0·01
1 Dec 1999	8·2	8·3	194	>0·05
26 Jan 2000	8·0	8·4	152	>0·05
Sprattus sprattus				
2 Nov 1999	6·5	6·7	234	<0·05
1 Dec 1999	6·4	6·7	126	<0·05
26 Jan 2000	6·4	6·5	174	>0·05

AFD, acoustic fish deterrent.

of sound through water. It can thus be expected that the shape of the sound field near the intake will also undergo some modifications with changing salinity and temperature but the modifications are not of such extent that they influence the sound system's efficiency.

Analysis of L_T data was performed on herring and sprat (Table III). The average size of both species decreased during sound emission but this decrease was limited to a few mm.

DISCUSSION

Artificial sounds have previously been used to reduce fish impingement by power plant intakes or turbines. Infrasound (10 Hz) produced flight and avoidance responses in Pacific juvenile salmonids (Knudsen *et al.*, 1997). Haymes & Patrick (1986) used low-frequency sound to frighten alewives *Alosa pseudoharengus* (Wilson) away from an experimental structure. They reported a reduction of between 71 and 99%. Clupeids can also detect ultrasound (Mann *et al.*, 1997) and high frequency sounds have been applied to reduce the catch of alewives at a power plant on Lake Ontario (Ross & Dunning, 1993). The present results clearly demonstrate that an AFD system producing low frequency sounds between 20 and 600 Hz can repel many fish species at an estuarine cooling water intake. The response to sounds and the associated reduction in numbers captured by the intake, however, appeared to be species-specific, varying from no effect to highly efficient deflection. The difference between species in intake rates between test conditions and control samples is attributed to species-specific differences in both sound detection, appropriate response and swimming performance.

Different avoidance reactions of species to the sound system are due to differences in hearing abilities, and in particular, to differences in auxiliary anatomic structures that improve the sensitivity to sound (Hawkins, 1986;

Helfman *et al.*, 1997; Popper & Lu, 2000). Sound propagates rapidly and efficiently through water. Sound sources set up a wave that travels through the water. The wave can be characterized at a particular point by the particle velocity, a 'to-and-fro' motion of the component particles, and the sound pressure, a variation in pressure above and below the ambient level. The 'to-and-fro' displacements that constitute sound are extremely small, of the order of nanometres (Hawkins, 1986). The motion of particles is detected in the inner ear.

Species with swimbladders, although not all, exhibited a clear avoidance behaviour to the AFD system as their intake numbers during sound emission significantly decreased. Significant results were obtained for the smelt and various acanthoptherygian taxa (*e.g.* gobies, perch, sea bass and sand smelt). In gadoids, the swimbladder is positioned closely to the inner ear, which makes gadoids sensitive to sounds. Trials of a similar AFD system at Hartlepool power station (Tees Estuary, U.K.) confirmed that gadoids were deflected effectively as the catch of whiting *Merlangius merlangus* (L.), which ranked third in cooling water samples, decreased by 53·5% (A.W.H. Turnpenny, J.M. Fleming, K.P. Thatcher & R. Wood, pers. comm.). At Doel, the contribution of gadoids to the annual catch is marginal, although 0+ and 1+ year whiting and cod *Gadus morhua* L. captures peak during November and December in fyke nets at a nearby mudflat (Maes *et al.*, 1997). During the present study, only 13 whiting were caught of which 10 were captured during 'sound off' conditions. Some species, although having a swimbladder, remained unaffected by the sound system. Representatives of the Gasterosteiformes (pipefishes and sticklebacks) have armoured bodies consisting of dermal plates that possibly decrease the effect of sound pressure waves on both the inner ear and the swimbladder.

Small movements of the swimbladder due to sounds can be transmitted to the inner ear directly through the tissue but also *via* anatomical structures. Such structures considerably increase the hearing abilities in fishes. Herring, sprat and anchovy *Engraulis encrassicolus* (L.) (Clupeiformes) have thin, hollow ducts extending anteriorly from the swimbladder that expand into gas-filled bullae in the inner ear (Helfman *et al.*, 1997). This connection increases the sensitivity of clupeoids to sound and explains the high avoidance reaction in herring and sprat to the acoustic deterrence system. In ostariophysan fishes such as the cyprinids, weberian ossicles connect the swimbladder to the perilymph of the inner ear resulting, again, in increased sensitivity for low frequency vibrations. White bream, however, was the only cyprinid to be significantly deflected by the system whilst other cyprinids showed no reaction or occurred in low numbers. In contrast to the present results, impingement rates of cyprinids at a river pumping station in York, England, were significantly reduced by 80% when sounds were produced to deter fishes (A.W.H. Turnpenny, J.M. Fleming, K.P. Thatcher & R. Wood, pers. comm.).

The presence of hearing-related modifications does not appear to guarantee successful deflection. Considerable differences in the reduction occurred amongst species with high sensitivity to sound that could not be explained as a function of their hearing abilities alone. Such fishes may well detect the AFD sounds, but if the current velocity towards the intake exceeds the maximum swimming speed, they cannot necessarily escape and become impinged. This is particularly true at cooling water intakes in estuaries where strong tidal currents probably prevent fishes of perceiving a current towards the intake unless they are nearly on the intake. The average current velocity of the river, caused by the tides, is $0{\cdot}65\,\mathrm{m\,s^{-1}}$. Given the surface area of the intake aperture and the cooling water flow and using hydrodynamics equations, model results (unpubl. data) show that the current velocity towards the cooling water inlet due to cooling water abstraction is $0{\cdot}52\,\mathrm{m\,s^{-1}}$ upon the intake itself and rapidly decreases to a few $\mathrm{cm\,s^{-1}}$ at 10 m off the intake (unpubl. data). If fishes detect the intake by entering the sound field, they can react by swimming away against the tidal current, and eventually also against the intake current. Based on swimming performance, it is argued that larger-sized individuals and species have a better chance to avoid capture than small-sized species and larval or juvenile life-history stages. This is supported by observations of smaller-sized herring and sprat in the samples taken when the AFD system was operating. Additionally, species with carangiform locomotion using trunk and tail have considerably higher maximum swimming speeds than species with an anguilliform type of locomotion or species using only fins for propulsion. Consequently, fishes with moderate to slow maximum swimming speeds such as eel *Anguilla anguilla* (L.) as well as small-sized life-history stages of goby species may still be impinged in reasonable numbers although it can be assumed that they are well able to detect sounds.

The reliability of extrapolating these results to other power station cooling water intakes seems relevant. Trials of the same system at the Hartlepool power station resulted in similar results as reported in this paper. The highest effectiveness was for clupeids (60–80%). The intake rate of other species possessing a swimbladder was lower (54%) while the catch of non-swimbladder species was reduced by only 16% (A.W.H. Turnpenny, J.M. Fleming, K.P. Thatcher & R. Wood, pers. comm.). It should be kept in mind, however, that the overall reduction rate will probably depend on local conditions as well, such as cooling water entrance speeds, background noise and ichthyofaunal composition of the surrounding waters.

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