Written Submission

for the

Royal Society for the Protection of Birds

Response to the Examining Authority's First Written Questions (ExQ1)

Submitted for Deadline 3

22 January 2025

Planning Act 2008 (as amended)

In the matter of:

Application by Morecambe Offshore Wind Limited for an Order Granting Development Consent for the Morecambe Offshore Wind Farm

Planning Inspectorate Ref: EN010121

RSPB Registration Identification Ref: 20049971

Contents

1. Introduction

1.1. The RSPB’s responses to the Examining Authority’s First Written Questions (ExQ1) are set out in the table below.

ExQ1	**Question to:**	**Question**	**RSPB response**
Biodiversity, Ecology and Marine Processes (BEM)			
1BEM46	The Applicant JNCC NE NRW NatureScot DAERA RSPB North West Wildlife Trusts	**Assessments** In paragraph 62 of the Offshore Ornithology Technical Note 1 (EIA) [REP1-080] it is noted that the NE advice in relation to the CEA was not to include historic projects with limited (or no) overlap with the construction and operational timeframe of the Proposed Development. (a) However, would the existing background mortality rates include those associated with these windfarms? If so, does there need to be an associated assessment from the removal of their effects as they are decommissioned? It is appreciated that the assessment is precautionary, but without removing any such effects, is there a risk that the assessment becomes over-precautionary, leading to mitigation that is not required? It is also appreciated that there is a separate discussion in relation to when the Barrow windfarm is to be decommissioned (see ExQ1GEN10) which may also need to be considered. This argument, taken to its logical conclusion, should also factor in any effects associated with the decommissioning of other windfarms (see Table 5.1 of Applicant's response to Actions from PM and ISH1 [REP1-085]) for longer-term effects).	The RSPB considers that until such time as there is legal certainty that an individual wind farm will be decommissioned by a specific date, then it is appropriate to (i) assume they will continue to operate regardless of any nominal decommissioning date and therefore (ii) continue to include them in cumulative effects assessments and in-combination assessments. Unless that legal certainty on decommissioning is available, then it is possible that an individual operator may seek to extend the lifetime of its asset for commercial reasons. We note from question GEN10 that there are differing views as to whether any such extension in operating period would require further consent. Therefore, it is the RSPB's view that the current approach to assessment applies an appropriate level of precaution.

ExQ1	Question to:	Question	RSPB response
		(b) Could the Applicant, JNCC, NE, NRW, NatureScot, DAERA, the RSPB and the North West Wildlife Trusts please give their views as to how the effects of the decommissioning of existing windfarms should be considered to avoid over-precautionary mitigation/ compensation	
1BEM47	The Applicant JNCC NE NRW NatureScot DAERA RSPB North West Wildlife Trusts	**Base cases** The ExA understands that, following NE advice, consented turbine parameters have been used as opposed to as built parameters on the basis that it is, theoretically, possible that the remainder of the consented scheme could be built out. (a) However, either where a scheme is coming to end of its life (see Table 5.1 of Applicant's response to Actions from PM and ISH1 [REP1-085]) or where the scheme as built would prevent additional development, should not 'as built' data be utilised? Would this alter any of the effects assessed? (b) Could the Applicant, JNCC, NE, NRW, NatureScot, DAERA, the RSPB and the North West Wildlife Trusts please give their views on this proposition.	In respect of a scheme coming to the nominal end of its life, we refer to our answer to 1BEM46 above. In respect of the question as to whether projects in the in-combination assessment that have been built out to a lower capacity than that consented should be assessed as "as built", we set out our view below. This may be acceptable for windfarms where the Development Consent Order (DCO) has been amended and therefore there is legal certainty regarding the reduction. However, where windfarms still have their original DCOs and therefore the ability to construct more wind turbines, it is not appropriate to do anything less than consider the full extent of those DCOs when considering in-combination/cumulative effects. To do otherwise risks underestimating the potential effects on important seabird populations and other environmental factors. The RSPB has no information to hand to be able to comment on whether there are any schemes "as built" that would prevent additional development. That would require a scheme by scheme assessment.

ExQ1	Question to:	Question	RSPB response
1BEM48	The Applicant JNCC NE NRW RSPB North West Wildlife Trusts	**Assessments** The Offshore Ornithology Technical Note 3 (Red-Throated Diver at Liverpool Bay SPA Update Assessment) [REP1-082] notes the effects of existing disturbance by helicopters and seacraft. It is stated that, apart from ferries, a significant proportion is associated with the oil and gas industry. As it well known, the decarbonisation agenda will mean that these operations will be phased out over time (re-purposing for Carbon Capture Assessment would need a revised assessment as it is not currently consented). Should, therefore, the effects of the removal of this traffic form part of the overall assessment? Could the Applicant, NE, NRW, the RSPB and the North West Wildlife Trusts please give their views on this proposition?	The same principles apply to the decommissioning of oil/gas platforms as to offshore wind farms that we set out in our answer to 1BEM46. At this stage we cannot have legal certainty that the disturbance caused by helicopters and seacraft associated with the oil and gas industry will cease or reduce in intensity by any specific date. Until such legal certainty is provided, it is appropriate to include the impacts of this traffic in the overall assessment.
1BEM50	RSPB The Applicant	**Manx Shearwater – disorientation due to lighting** The RSPB challenges the assessment of no adverse impacts on Manx shearwater through collision with rotating turbines, highlighting concern about disorientation of shearwaters from lighting. It cites publications relating to collisions with lighthouses and other illuminated structures. The Applicant's 'Response to Relevant Representations' [PD1-011] references other papers, which present a counter view (eg at RR-073-13). Could the RSPB and the Applicant submit these papers into the Examination.	Please see Annex A to this submission for copies of the requested references. Archer, M., Jones, P. H., & Stansfield, S. D. (2015) Departure of Manx Shearwater Puffinus puffinus fledglings from Bardsey, Gwynedd, Wales, 1998 to 2013 *Seabird, 48* 43-47 Guilford, T., Padget, O., Bond, S., & Syposz, M. M. (2019). Light pollution causes object collisions during local nocturnal manoeuvring flight by adult Manx Shearwaters Puffinus puffinus. *Seabird*, *31*

ExQ1	**Question to:**	**Question**	**RSPB response**
			Miles, W., Money, S., Luxmoore, R., & Furness, R. W. (2010). Effects of artificial lights and moonlight on petrels at St Kilda. *Bird Study*, *57*(2), 244-251
Habitats Regulations Assessment (HRA)			
1HRA38	The Applicant RSPB MMO	**Ecosystem effects due to ocean stratification** The RR from the RSPB [RR-073] references the ecosystem impact of water column stratification on prey availability. The Applicant's comments on WR at D2 item WR-112-11 [REP2-027] suggests that this issue may have been resolved in SoCG discussions with MMO. (a) Is the RSPB able to provide specific evidence to demonstrate that such an effect is likely for example, the provision of the Isaksson et al (2023) reference, where relevant? (b) The Applicant's response to RR item RR-073-16 [PD1-011] responds to the RSPB comments, cross referencing ES Chapter 12 [REP1-032]. Neither of the cross-referenced sections of text explicitly address stratification. (c) Can the MMO confirm that it is satisfied with the Applicant's approach to consideration of water column stratification? (d) Could the Applicant please address this point (it is noted that ES Chapter 7 [REP2-008] does include reference to stratification	Please see Annex B to this submission for a copy of the following reference relevant to the ecosystem impact of water column stratification on prey availability. Isaksson, N., Scott, B.E., Hunt, G.L., Benninghaus, E., Declerck, M., Gormley, K., Harris, C., Sjöstrand, S., Trifonova, N.I., Waggitt, J.J. and Wihsgott, J.U., 2023. A paradigm for understanding whole ecosystem effects of offshore wind farms in shelf seas. *ICES Journal of Marine Science*, p.fsad194.

Annex A: references requested in Examining Authority question 1BEM50

Archer, M., Jones, P. H., & Stansfield, S. D. (2015) Departure of Manx Shearwater Puffinus puffinus fledglings from Bardsey, Gwynedd, Wales, 1998 to 2013 *Seabird, 48* 43-47

Guilford, T., Padget, O., Bond, S., & Syposz, M. M. (2019). Light pollution causes object collisions during local nocturnal manoeuvring flight by adult Manx Shearwaters Puffinus puffinus. *Seabird*, *31*

Miles, W., Money, S., Luxmoore, R., & Furness, R. W. (2010). Effects of artificial lights and moonlight on petrels at St Kilda. *Bird Study*, *57*(2), 244-251

Departure of Manx Shearwater *Puffinus puffinus* fledglings from Bardsey, Gwynedd, Wales, 1998 to 2013

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Very little has been published about how soon Manx Shearwater *Puffinus puffinus* fledglings leave the area around the natal colony, clearly because of the difficulty of re-catching them after they are first ringed and of knowing whether or not recaptured birds had flown (Perrins *et al*. 1973; Perrins 2014). Fledglings are thought to depart southwards promptly after they finally leave the natal burrow because the food supply near the colony is deteriorating as the breeding season draws to a close, as suggested by the earlier departure of the adults and the lighter weights of later fledglings, and also as evidenced by the paucity of recoveries immediately after fledging, gale-wrecked birds apart (Brooke 1990). Catching Manx Shearwaters on Bardsey, Gwynedd, Wales at lighthouse attractions within the perimeter of the lighthouse complex, and re-catching them there and on the ground elsewhere on the island, showed that not all fledglings left the area immediately, with a few remaining for up to five/six days. The data published here relate to the numbers initially attracted to the lighthouse and later re-attracted or otherwise caught again in late August and September, between 1998 and 2013.

Bardsey (52°76’N 4°78’W) is three km long from north to south and a little over one km at its widest point (Figure 1). A small colony of breeding Manx Shearwaters has been known on the island since early in the 20th century (Cramp *et al*. 1974) and the species currently breeds in burrows, mainly in soil on the steeper slopes, but also in or by the earth-covered stone boundary walls in the low-lying areas. The island’s Manx Shearwater colony is perhaps the fifth largest in Britain and Ireland (Newton *et al*. 2004) and the most recent whole-island surveys in 2001 (Leaper 2001; Newton *et al*. 2004) and in 2008, 2009 and 2010 (Else 2009, 2010, 2011) estimated the breeding population at between 9,000 and 16,000 pairs. The productivity mean for the ten-year period 2001–10 was 0.77 (Brown & Stansfield 2011). In 2008–10 the island was subdivided into three survey areas for censusing the population: Area A on the steep east side cliffs held 9,610 apparently occupied burrows (AOB), Area B held 4,071 AOB, and Area C held 2,178 AOB (Else 2009, 2010, 2011). The lighthouse stands in Area C towards the southern tip of the southern promontory (Figure 2), at a point where burrow densities are low, there being probably fewer than 50 AOB within a radius of 100 m.

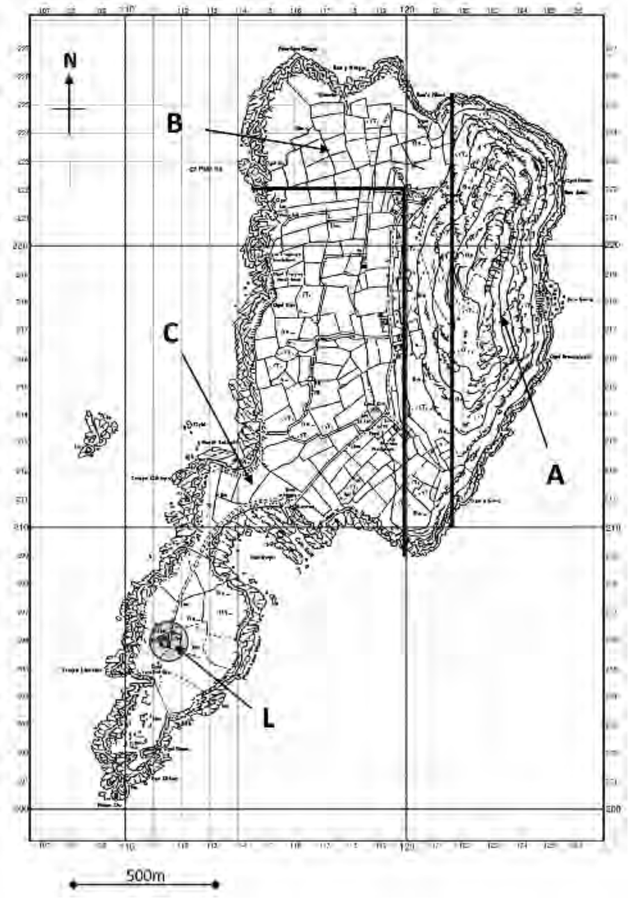

Figure 1. Map of Bardsey showing the position of the lighthouse (L) and the extent of the three survey areas (A, B and C).

Juvenile shearwaters exercise their wing muscles vigorously outside the burrows during the final nights before fledging (Brooke 1990), but very few of these on Bardsey were observed to fly, and those that did flew no more than 20–30 m in level flight nor to a height of more than one metre. The lighthouse lantern light is at about 30 m above ground level, and during 1998–2013 birds were attracted to the light in numbers in weather conditions involving very poor visibility and general mistiness, with rain and a cloud ceiling not far above the upper part of the lighthouse; very small numbers were also attracted from time to time on clearer nights. The attracted birds usually circled in the revolving beams of the light for several minutes and often much longer before colliding with the light and falling to the ground or escaping its attraction and flying off (Stansfield 2010). Birds found on the ground within the lighthouse complex can be assumed to have been free-flying, as a substantial perimeter wall prevented fledglings from wandering into the complex area on foot (Figure 3). Further, 27 of the fledglings attracted, but not re-attracted, were retraps, of which 23 had been ringed as juveniles at least 600 m and as much as 2,500 m from the lighthouse, and between two and 13 nights earlier (mean = 4.91). Thus juveniles attracted to the light were assumed to be fledglings and on their maiden or a subsequent flight. Trinity House altered the light in 2014, precluding the likelihood of any further attractions and gathering further like data.

Regular checks were made at night by Bardsey Bird Observatory staff for attracted birds, which were later ringed. Juveniles were distinguished from adult birds, often by retained down on the crown and/or belly, but also by a combination of fresh, unworn primaries with no bleaching on the tips of the primaries or secondaries, unworn and hence sharp claws, and more subjectively, the soft juvenile feel and, on occasion, the characteristic earthy burrow-smell. None showed any sign of injury.

Whilst the precise time of capture at attractions was not recorded, the post-midnight date was ascribed by convention to the ringing procedures, regardless of whether a bird was picked up before or after midnight. These birds were usually ringed, if not previously ringed at the burrow, (processed) and released by manual launching into the wind, to fly away over the sea and out of sight, in mid/late afternoon or in the evening before dusk, and therefore possibly up to 20 hours after capture, and possibly within four hours of re-capture on the 'following night'. The birds picked up elsewhere and subsequent to the initial attraction were processed there and then and the date and time were usually recorded to the nearest hour.

Accepting that the attracted fledglings were almost certainly reared on Bardsey, at least 21 of those first attracted did not permanently leave the immediate vicinity of their natal colony for between one and six days, 16 being re-captured at lighthouse attractions and five on the ground elsewhere on the island (Table 1). Of those five, three were at least 750 m distant from the lighthouse at the West Coast (1) and in the Lowlands (2), one was c. 600 m distant at Solfach, and the fifth was at the South End, between 50–400 m distant. These 21 fledglings were part of a cohort of 220 fledglings caught at attractions on ten nights. To complete the picture, a further 628 fledglings, caught below the lighthouse on 177 nights at other very minor attractions and usually in twos and threes, were not caught again. The attracted birds were assumed to be Bardsey-fledged. Whilst the possibility of catching a fledgling from a colony elsewhere cannot be ruled out entirely, only a single fledgling ringed elsewhere has been controlled on Bardsey since 1953, compared with 140 adults. Furthermore, the likely minimum number of fledgling shearwaters departing Bardsey in any of the subject years would be about 6,900, calculated by multiplying the likely minimum number of breeding pairs (say 9,000) by the likely productivity mean (0.77).

Figure 2. View of Bardsey, looking southwest towards the lighthouse from the slopes of Mynydd Enlli, May 2015. © *Connor Stansfield*.

Figure 3. Bardsey Lighthouse and the perimeter wall that surrounds it, viewed from the south, May 2015. © *Connor Stansfield*.

Table 1. Numbers of fledgling Manx Shearwaters *Puffinus puffinus* attracted (A) at the Bardsey lighthouse and re-captured at a subsequent attraction or elsewhere on the island shortly afterwards, 1998–2013 and (B) captured and ringed on the ground on Skokholm and re-captured there afterwards, 1967, and the interval in days between captures.

Number of days from first to last capture	A. Number of Bardsey recaptures	Total days: product of first and second columns	B. Number of Skokholm recaptures	Total days: product of first and fourth columns
1	6	6	22	22
2	11	22	23	46
3			14	42
4	2	8	5	20
5	1	5	3	15
6	1	6	2	12
7			2	14
8			1	8
Totals	21	47	72	179
		Mean = 2.24		Mean = 2.49

The Skokholm numbers given here are slightly at variance with those in Perrins *et al.* (1973); of the 72 birds, 39 were weighed on both first and last recapture, covering a total of 104 'days' with a mean loss per day of 14.4 g.

Further, the birds caught at the first attractions may or may not have been on their maiden flights and there is no sure means of telling. As the majority of Welsh birds go overland to the sea on fledging (Brooke 1990), rather than flying, those scrambling to the sea might not have had time, or been able to fly well enough, to be attracted to the lighthouse light on their first night at sea, thus extending the post-fledging periods before the recapture attractions or other recapture for at least another 24 hours and perhaps longer.

So infrequent were the closely-spaced nights of double attractions that only 16 fledglings were both attracted and re-attracted to the lighthouse, while five originally-attracted birds were subsequently recaptured on the ground elsewhere on the island during occasional general ringing activities. The recapture of these five attracted fledglings, away from the lighthouse, indicates that some birds return to land after maiden fledging or other flights and before final departure.

Similar, but larger, samples of data arise from the capture, ringing and recapture of Manx Shearwaters on the ground outside the burrow on Skokholm Island, Pembrokeshire, Wales in 1967 (Table 1). The date of ringing on the surface was taken as indicative of the date of fledging in respect of birds not assumed to have ever flown (Perrins 2014).

Acknowledgements

Thanks are due to Professor Chris Perrins, the Wildlife Trust for South & West Wales (its current name) and Skokholm Bird Observatory for providing the Skokholm data in Table 1 with commentary. Members of the Bardsey Bird Observatory staff and visitors caught, re-caught, ringed and processed the fledglings at attractions and elsewhere on the island. Martin Heubeck, Professor Chris Perrins and another, anonymous, referee greatly improved the manuscript with their comments.

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Full title: Light pollution causes object collisions during local nocturnal manoeuvring flight by adult Manx shearwaters (*Puffinus puffinus*)

Running head: Lights out for Manx shearwaters

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Abstract

Understanding the detrimental effects of anthropogenic light on nocturnally mobile animals is a long-standing problem in conservation biology. Seabirds such as shearwaters and petrels can be especially affected, perhaps because of their propensity to fly close to the surface, making them vulnerable to encountering anthropogenic light sources.

We investigated the influence of light pollution on adult Manx shearwaters at close range in foggy conditions. We recorded collisions with a building at a breeding colony for six consecutive pairs of intervals in which the house lights were left on as normal for 135 seconds, then turned off for 135 seconds. The relationship between lighting condition and collision frequency was highly significant, with a collision rate in the presence of lighting around 25 times that in its absence. Our results show that birds were clearly affected by the lights, by being either directly attracted, or disorientated during flight close to the structure. This could have been due to the light source itself, or an indirect effect of the all-round reflective glow in the fog perhaps interfering with visual or magnetic control inputs on both sides of the bird simultaneously. Our results suggest a mechanism by which the screening of artificial lights close to shearwater breeding areas, at least during foggy nights, could lead to improved welfare and

survival at breeding colonies.

Introduction

Understanding the detrimental effects of anthropogenic light on nocturnally mobile animals is a long-standing problem in conservation biology (Montevecchi 2006; Gaston *et al.* 2013; Gaston *et al.* 2014). Artificial light at night can draw individuals from long distances, whilst repelling others, affecting many activities including foraging (e.g. Garber 1978; Frank 2009; Pereszlényi *et al.* 2017), reproductive behaviour (e.g. Miller 2006; Kempenaers 2010; de Jong *et al.* 2015; Russ *et al.* 2017), and daily, monthly or annual movements (e.g. Salmon 2003; Mathews *et al.* 2015; Rodríguez *et al.* 2017).

Amongst the most endangered groups of birds (Croxall et al. 2012), shearwaters and petrels (Procellariiforms) are especially badly affected by anthropogenic light sources and the grounding of their fledglings in particular has been studied in many parts of the world (Rodríguez *et al.* 2017). Manx Shearwaters (*Puffinus puffinus*), which breed at island colonies predominantly around the UK and Ireland, are classified as of Least Concern by the IUCN, but still they are often reported grounding in artificially lit urban areas or at other light sources particularly during the fledging period (Brooke, M. 1990; Le Corre *et al.* 2002; Rodríguez et al. 2008; Miles *et al.* 2010; Archer *et al.* 2015). There are fewer studies on the grounding of adults on land, in this or other species, since adults usually constitute a small percentage of the individuals affected in any fallout (Le Corre *et al.* 2002; Rodríguez & Rodríguez 2009; Telfer *et al.* 1987). Here we report a short experiment designed to test the instantaneous effect of anthropogenic house light on collisions with a man-made object, and outside of the fledging period when only adults are present at breeding colonies.

Electric lighting is used to provide night service to the seasonal residents (a nightly summer maximum of 42 staff and tourists, of which up to 10 might stay in the Island Office) of the several buildings on Skomer Island National Nature Reserve, which is an internationally important Manx shearwater breeding colony. Shearwaters returning to the colony sometimes collide with the buildings and higher numbers of crashing seabirds are expected on cloudy and rainy nights (Telfer *et al.* 1987; Jones 1980). It may be that more birds visit breeding colonies in such conditions anyway, when there is less ambient light from the moon (Riou & Hamer 2008), but it is also possible that visual guidance in local manoeuvring flight is less effective and this contributes to collision risk. To determine whether there is a local effect of artificial light on collisions with the structure from which it is emanating, we conducted a very short experiment.

Study area

Skomer Island (51° 44´ N; 5° 19´ W) hosts the biggest colony of Manx shearwaters in the world, an estimated 316 000 breeding pairs, making up around 36% of the global breeding population when combined with the neighbouring islands of Skokholm and Middleholm (Perrins *et al.* 2012), The other species active at night on the island include European storm petrels (*Hydrobates pelagicus*), rarely seen close to the Island Office and not on the night of our experiment, and extremely unusually, vagrant shearwater species. There are no migratory passerines at this time of year. By contrast, thousands of Manx shearwaters fly low over the colony each night at this time of year as they return to their burrow nests (if breeding), or in display flights whilst calling to prospective partners. Typically, when a shearwater collides with the building it will make a loud thud (too loud for a smaller bird), and will fall to the ground where it will remain for a few seconds or minutes, appearing stunned, before walking into the undergrowth. It is therefore overwhelmingly likely that every single impact we heard was of a different individual Manx shearwater. We do not know the fate of crashed birds in general, but very occasionally we find a bird killed by the impact or bleeding from the head (TG personal observations), suggesting that whilst the vast majority escape to cover in the minutes following collision, there is the potential for serious injury which might affect future survival. We did not attempt to recover or assess birds for the effects impact during this experiment.

Methods

On 27th May 2015, whilst undertaking fieldwork, we noticed that for several hours there were many shearwater collisions with the building. We therefore opportunistically conducted an experiment, exploiting the occasion of an unusually foggy night. The number of significant shearwater collisions with a prominent man-made structure, the Island Office (Fig. 1), on Skomer Island NNR, was recorded by tallying in a notebook each audible crash with roof, walls or windows that could be heard from inside the researchers' quarters located centrally on the seaward (north) side of the structure. The predominantly wooden building is of a two-part design with a lower one-storey section measuring 17.5m x 7.7m x 5.5m

high at the top of the pitched roof leading to a taller two-storey section measuring 7.6m x 12.0m x 8.5m at the highest point. The design of the study was decided as soon as we started observing the phenomenon and data on collision incidents were then recorded for six consecutive pairs of intervals in which (A) the house lights were left on as normal for 135 seconds, then (B) the lights were turned off. This interval (2.25 minutes) was decided arbitrarily but to provide a short period (half an hour) that would allow six replicates during the middle of the night whilst the weather conditions persisted and before the colony became empty as the nights in May are short at this latitude. No other artificial lighting, except the very dim emergency exit panels and LEDs from electronic equipment indoors, was visible in the building. Luminance from two white fluorescent twin

tube compact lights (Pro-light 11W 2700k) was visible externally via two rectangular glass windows (each 0.8m x 1.0m tall), and one small (0.37m diameter) round window in the external door, and it is these sources that were extinguished during the lights-off treatment. The experiment started at 2330 GMT and lasted 27 minutes.Results

The relationship between lighting condition and collision frequency was highly significant (Wilcoxon Test W=0, N=6, P=0.0044; Table 1), with a collision rate in the presence of lighting (5.9 collisions per minute) around 25 times that in its absence (0.2 collisions per minute).

Table 1. Results from study shows the collision count data (rates per minute in brackets) across the six sequential treatment interval pairs.

Interval	**I**	**II**	**III**	**IV**	**V**	**VI**	**Mean**	**Median**
Lights on	13 (5.8)	14 (6.2)	10 (4.4)	11 (4.9)	18 (8.0)	14 (6.2)	13.3 (5.9)	13.5 (6.0)
Lights off	1 (0.4)	0 (0)	0 (0)	1 (0.4)	0 (0)	1 (0.4)	0.5 (0.2)	0.5 (0.2)

Discussion

The extinguishing of artificial building lights caused a dramatic (25-fold) and almost instantaneous reduction in shearwater collisions with the man-made structure from which the light was emanating. It is possible that the sound of one bird may be masked by another, so our numbers may even be, to some extent, underestimates. In addition, the recorders informally observed that collisions during the lights-off treatment always occurred shortly after the switch in treatment, suggesting that collisions during the lights-off period were mainly influenced by the effect of the light that had just been turned off. Because the treatment intervals were short (135 seconds) our experiment suggests a predominantly local effect of lighting, with collisions by birds already at the colony and in the vicinity of the buildings. In an earlier GPS tracking study we estimated the mean speed of Manx shearwaters in flight to be around 11m.s^{-1} (Guilford et al. 2008), allowing for the maximum attraction distance during an illuminated period to be about 1.5km on this relatively calm night. Of course it is possible that birds may have been attracted in to the area by illumination prior to the experiment, or during a previous lights-on interval during the experiment itself, but restricted penetration of light in foggy conditions is likely to mean that in fact our building lights were only visible from much shorter distances. Furthermore, the often immediate effect of turning on the lights and the striking disparity in collision rates between the on and off treatments, strongly suggests that a local effect of the light is responsible for inducing collision. A different design would be required to determine whether, as has been suggested for urban groundings (e.g Miles *et al.* 2010; Reed *et al*. 1985), artificial lights can also attract birds from longer distances under some conditions. A study using GPS on Cory's shearwaters Rodríguez *et al.* (2015) showed that locations where birds were rescued had greater light pollution levels than at colonies, and found that areas with high intensity light attracted birds from further away than areas with low intensity light. In our study, birds already at or close to the colony are either being attracted by the light source locally, or are being disoriented during visually guided flight close to the structure either directly by the light source or indirectly by the reflected glow in the fog.   These two effects could operate in combination: Day *et al.* 2003). Although it might have been interesting to replicate this experiment under different conditions this has not been possible because curtains have been fitted to the building windows (in response to our observations) so that very restricted light now emanates from the structure under normal operation. Paradoxically, manipulation of light levels would now involve knowingly inducing potentially fatal collisions with the structure, whereas in the current experiment it opportunistically involved reducing them, thereby changing the ethical and legal basis of the research under UK law.

The mechanism by which light interferes with normal behaviour in flying birds is unknown. Hypotheses range from direct interference with stellar, lunar, or magnetic compass orientation mechanisms, to a normal attraction to light sources having its origin in mechanisms for hunting bioluminescent prey (for reviews see Montevecci 2006; Gaston *et al.* 2013). In our experiment birds would have been engaging in control flights above the colony rather than directional flights of any distance. This makes it unlikely that the mechanism interfered with was part of either long-distance guidance (the role normally assumed for compass orientation in migratory birds for example), or hunting. Direct attraction to the light is a possibility (Reed, 1986), but it is interesting that most of the collisions witnessed in our experiment were not with the windows themselves, from which the light emanated, but with the surrounding structure. Furthermore, in fog as here light is scattered to produce a locally bright glow but with little distance penetration, so birds should if anything be attracted from less far than on clear nights. Although we did not explicitly compare fog with clear conditions in this experiment, it is certainly the informal experience of researchers (see also Black, 2005) that collisions are much rarer on clear nights which is the opposite of what would be expected if direct attraction was responsible. In the absence of the light, birds are apparently normally able to avoid collision with this structure despite the poor visibility. One hypothesis, therefore, is that sudden proximity to relatively bright light may disrupt the ability of birds to use their normal dark-adapted visual guidance effectively. This could be more pronounced on dark nights when birds are more dependent on dark-adaptation. Perhaps the scattering of light in fog also contributes by interfering with dark-adaptation in multiple directions, simultaneously disabling low-light visual guidance input from both eyes. A second hypothesis, however, is that birds manoeuvring in low light conditions might make use of a magnetic compass as a “heading indicator” (Guilford & Taylor 2014) to monitor and control local changes in orientation relative to the ground. The suggestion that birds moving in a fluid medium might use compasses as heading indicators in flight control has been made before for a sun-compass during diurnal flight (Guilford & Taylor 2014), and we now know that Manx shearwaters have a time-compensated sun compass (Padget *et al*. 2018), but in nocturnal flight when solar cues are unavailable a magnetic compass might operate in a similar way. It is possible, therefore, that a light-dependent magneto-receptor (Hore & Mauritsen 2016) becomes temporarily disrupted by saturation in the presence of bright light, and that this disrupts the bird’s ability to guage its heading changes during local flight manoeuvers, causing collision. Again it is possible that in fog input from both eyes (eyes are thought to be the organs responsible for sensing magnetic direction) become affected simultaneously because of local light scattering.

Whether the effect of light witnessed in our experiment is an effect on visual or magnetic guidance in flight control remains to be determined. However, our results do suggest that light pollution may cause interference effects at several scales, with disruption of local flight control in addition to one or more attraction or compass disorientation effects operating at longer distances.

Measurements made during a rescue programme for Manx shearwaters suggested that around 7% of fledglings die as a result of grounding (Syposz *et al*. 2018), but such estimates are biased and the likely death rate may generally be much higher (about 40% in a study of short-tailed shearwaters) where humans do not intervene (Le Corre *et al.* 2002; Rodríguez & Rodríguez 2009; Fontaine *et al.* 2011; Rodríguez *et al.*, 2017). In particular the seriousness of the effect on flying shearwaters of collision with a man-made structure is not well understood.

Shearwaters rarely alight on a terrestrial surface with great control unless the wind strength and direction is very favourable (personal observation), so it is common to see or hear them crash into the undergrowth during normal attempts to land and they may be at least partially adapted to rough landings. However, after collision with buildings birds can usually be found sitting still for some time before making their way to cover, but on occasion may be found bleeding from the beak or killed by the impact. The longer-term effect of collisions on birds that survive immediate impact is not known, however. Birds that perish on the surface during the night, or fail to reach their nests, are likely to be removed and eaten by aerial predators and scavengers (mainly great black-backed gulls *Laurus marinus* on Skomer, where there are no mammalian predators) (Raymond *et al.* 1993).

Our results suggest that the normal controlled flight behaviour of adult shearwaters can be severely negatively affected by proximity to artificial lights on structures under some nocturnal conditions. They suggest a mechanism by which the screening of artificial lights close to shearwater breeding areas, at least during foggy nights, could lead to improved welfare and survival at breeding colonies. Off-shore, artificial light sources on structures or vessels are known to cause collisions in open water, particularly during foggy conditions, which can be reduced by light-screening or reduction (Black, 2005; Glass & Ryan, 2013). So in addition it is possible that lights on vessels close to colonies, or close to the flight paths of returning birds, might interfere with collision avoidance behaviour in adult shearwaters, even if they do not attract birds from a distance. Large, highly lit tankers commonly anchor in the waters close to Skomer, but their effect on the Manx shearwaters remains unknown.

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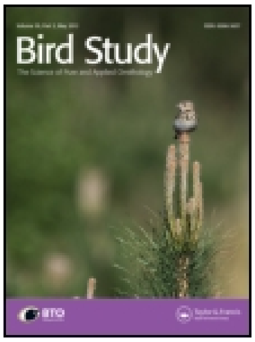
Figure 1. Photograph of the Island Office showing its position on the shearwater nesting slopes at north haven on Skomer island. The windows through which light emanated during the lights-on treatment are the two furthest right of the line of five in the lower section of the building, and a small circular window in the door immediately right of these.

Table 1. Results from study shows the collision count data (rates per minute in

brackets) across the six sequential treatment interval pairs.

Interval	**I**	**II**	**III**	**IV**	**V**	**VI**	**Mean**	**Median**
Lights on	13 (5.8)	14 (6.2)	10 (4.4)	11 (4.9)	18 (8.0)	14 (6.2)	13.3 (5.9)	13.5 (6.0)
Lights off	1 (0.4)	0 (0)	0 (0)	1 (0.4)	0 (0)	1 (0.4)	0.5 (0.2)	0.5 (0.2)



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Effects of artificial lights and moonlight on petrels at St Kilda

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Capsule When moonlight levels are low, shearwaters and storm-petrels are attracted to artificial lighting at night at St Kilda and may be killed, but impacts are lessened by deliberate light reduction measures.
Aims To determine the scale and impacts of attraction of petrels to artificial lights at St Kilda, investigate influences of the lunar cycle, and assess effects of reducing artificial light emissions.
Methods Nightly numbers of Manx Shearwaters *Puffinus puffinus*, Leach's Storm-petrels *Oceanodroma leucorhoa* and European Storm-petrels *Hydrobates pelagicus* attracted by artificial lights were recorded in September and October from 2005 to 2008. Effects of experimental reductions to light emissions in 2007 and 2008 were assessed, together with variation in annual moonlight, mortality rates, and age of birds found.
Results Reductions to light emissions caused a decrease in numbers of Leach's Storm-petrels attracted, but had less effect on attraction of Manx Shearwaters. Only juveniles were found, the majority after nights with little or no moonlight, and mortality was extremely infrequent. Only one European Storm-petrel was found, and Leach's Storm-petrel and Manx Shearwater totals were small compared with estimated breeding totals at St Kilda.
Conclusions Numbers of petrels attracted to artificial lights on St Kilda were low. However, reductions to light emissions were still beneficial in reducing numbers of young that became disorientated, grounded, or died during fledging periods. Therefore, reductions to light emissions should be encouraged. A review of this phenomenon across the UK found it to be rare in breeding areas away from St Kilda.

Attraction to artificial lights has been observed in many different species of birds (Saunders 1930, Herbert 1970, Avery *et al.* 1976, Dick & Donaldson 1978, Harris *et al.* 1998). Among seabirds, burrow-nesting and nocturnal species such as petrels (Procellariiformes) are particularly vulnerable (Klomp & Furness 1992, Jones & Francis 2003, Montevecchi 2006). Widespread mortality of petrels has been reported in many situations where these birds are attracted to artificial lights, especially on islands with large breeding populations of shearwaters, storm-petrels, and gadfly petrels (Reed *et al.* 1985, Muirhead & Furness 1988, Brooke 1990, Warham 1996, Brooke 2004, Imber *et al.* 2005, Montevecchi 2006, Salamolard *et al.* 2007, Rodríguez & Rodríguez 2009). Tens of thousands of light-disorientated and grounded petrels have been recorded, and many birds found dead, including threatened, endangered, and endemic species (Reed *et al.* 1985, Stewart *et al.* 1996, Jones 2001, Le Corre *et al.* 2002, Le Corre *et al.* 2003, Montevecchi 2006). On several islands, conservation measures have been implemented to reduce the impacts on petrels of artificial light from buildings and to decrease further threats to grounded petrels from mammalian predators (Le Corre *et al.* 2002, Montevecchi 2006). For example, on the island of Kauai, Hawaii, large numbers of Newell's Shearwaters *Puffinus newelli*, Madeiran Storm-petrels *Oceanodroma castro* and Dark-rumped Petrels *Pterodroma sandwichensis* have been attracted to bright lights of coastal resorts, but by shielding lights to prevent upwards radiation in the largest resorts, the number of birds attracted decreased by 40% (Reed *et al.* 1985). On Tenerife, Canary Islands, public awareness and civil cooperation with care and release schemes for petrels found around the heavily-lit resorts have resulted in the

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successful release to sea of 95% of nearly 10 000 petrels found between 1998 and 2006, including Cory's Shearwaters *Calonectris diomedea borealis*, Bulwer's Petrels *Bulweria bulwerii* and White-faced Storm-petrels *Pelagodroma marina* (Rodríguez & Rodríguez 2009).

In the UK at St Kilda, Outer Hebrides, Manx Shearwaters *Puffinus puffinus* and Leach's Storm-petrels *Oceanodroma leucorhoa* have been found grounded within the inhabited area of the only village, on the island of Hirta, annually since 1969 (*St Kilda Rangers' Reports* 1969–2004, St Kilda Rangers pers. comm.). Although records have not been systematically documented every year, it is certain that in excess of 10 000 shearwaters, storm-petrels, and Atlantic Puffins *Fratercula arctica* have been found, and that these species are strongly attracted in autumn to the lights of buildings at night in the village, and formerly to streetlamps that were on at night along the shorefront (*St Kilda Rangers' Reports* 1969–2008, Harris 1984, Harris *et al.* 1998, St Kilda Rangers pers. comm., Miles & Money pers. obs.). In a successful attempt by the Ministry of Defence (MoD) and The National Trust for Scotland (NTS) to reduce numbers of puffins found grounded on Hirta, these streetlamps were turned off by the St Kilda MoD base staff through the late 1990s, and have remained turned off to date. The village on Hirta faces the island of Dùn, across Village Bay, and it is assumed that young Manx Shearwaters and Leach's Storm-petrels attracted to the village lights are mostly fledglings from the large breeding colonies on Dùn, since these species do not breed on Hirta within sight of the village. Unlike puffins (Harris *et al.* 1998), storm-petrels and shearwaters attracted to artificial lights in the village on Hirta have not been studied and, until now, mortality rates, ages of all birds, and influences of the moon on the attraction of petrels to lights at St Kilda were unknown. Dùn holds the largest breeding colony of Leach's Storm-petrels in Britain and Ireland (Mitchell *et al.* 2004). Recent declines in this colony have been reported, from an estimated 27 704 apparently occupied sites (AOS) in 1999 to 14 490 AOS in 2003 and 12 770 AOS in 2006 (Newson *et al.* 2008). Predation of Leach's Storm-petrels by Great Skuas *Stercorarius skua* has been proposed as the most likely cause of a decline, but other possible mortality factors for petrels should not be ignored. This study aimed to assess the numbers, ages and mortality of petrels attracted to the lights in the village on Hirta between 2005 and 2008; to determine the possible mitigating effects of reduced artificial lighting in the village at night; and to investigate the influence of the lunar cycle on storm-petrels and shearwaters found on Hirta.

METHODS

Study site, species, and collection of grounded petrels

St Kilda (57°47′N, 08°33′W) is located in the Outer Hebrides 66 km west of Harris. This study was carried out on the largest island in the archipelago, Hirta, in the inhabited area (0.25 km^2) of Village Bay. Petrels found grounded were Leach's Storm-petrels, European Storm-petrels and Manx Shearwaters. No other species of petrel has ever been found grounded on St Kilda owing to light attraction, and these three are the only breeding petrels on the islands, other than Northern Fulmar *Fulmarus glacialis* (Murray 2002, Mitchell *et al.* 2004), which differs in not being an exclusively nocturnal visitor to land. In all years from 2005 to 2008, the entire perimeter of every inhabited building was systematically checked for grounded petrels, within the hour after dawn, every morning between 1 September and 16 October. These dates were chosen because the vast majority of grounded petrels found prior to this study had been recorded within this period (*St Kilda Rangers' Reports* 1969–2004). Searches also included thorough examination of all potential hiding places for grounded petrels, including pipe systems, nearby vehicles, and extraction vents. All birds examined were fledglings with newly grown fresh feathers and sometimes tufts of chick down still present. When estimating the ages of Leach's Storm-petrels reference was made to photographs of known adults examined during ringing and known juveniles from burrows, examined pre-fledging at St Kilda under license. Birds examined were temporarily sheltered in the dark and on the same day released to sea at dusk. Sheltering the birds prevented any chance of otherwise exposed individuals being found by skuas, which commonly hunted within the village area during daylight (Miles & Money pers. obs.). The timing of release aimed to minimize this threat, but also to reduce the likelihood of the birds flying back towards artificial lights in the village, which were much less glaring at dusk than later in the night.

Artificial lighting and reduction measures

In 2005–2008, total artificial lighting at night in the village on Hirta included: 32 fixed outside lights, indoor lighting permanently on in two utility buildings (for access safety), and indoor lights left on at night with windows uncovered in up to 15 rooms used for accommodation. Eleven buildings in the village were used or inhabited with lighting on during nights of this study, all but two being MoD buildings of the radar base facility. The small power station for the island was the most densely-lit building, with 24-hour indoor lighting and eight outside lights. In 2005 and 2006, many indoor lights in the village were left on at night, many left uncovered, outdoor lights left on, and no reductions to light emissions made. In 2007, at our request, measures to reduce light emissions to the absolute minimum in the village were kindly implemented by staff of the radar base and NTS. These changes included all outside lights being turned off and the windows of the majority of rooms being shielded at night by curtains, blinds, or custom-made boarding. However, effects of these measures on petrels were somewhat unclear from one year's trial (see Results). In 2008, light-reduction methods were repeated as in 2007, but with an experimental period of 20 nights of no light reduction in the village, starting from the night of 22 September. The exact start date was determined by the day most convenient to the radar base staff for changing all light reduction measures on their buildings, and because of this could not be chosen entirely at random (see Discussion). The timing and short duration of the 20-day control period were considered preferable to lights being left on and uncovered for the entire late summer and autumn in 2008, because attraction of fledgling puffins would be minimized in the late summer, and numbers of petrels attracted in different light conditions would potentially be comparable within years as well as between years.

Influence of moon phase and position

We investigated possible effects of moonlight on numbers of petrels found attracted to lights in the village using two explanatory variables: the phase of the moon and the length of time that the moon was above the horizon at night. Data of percentage of the moon's face illuminated (moon phase) and percentage total duration that the moon was above the horizon at sea level between sunset and sunrise were calculated for the years of this study using annual and daily data for St Kilda from the US Naval Meteorology and Oceanography Command (www.usno.navy.mil [accessed April 2009]). Effects of the moon on daily numbers of Leach's Storm-petrels and Manx Shearwaters found in the village were investigated using a GLM with a log-link function, and arcsine transformations for proportional data were used for moon variables. All analyses were performed using R version 2.8.1.

RESULTS

Numbers, ages, and mortality of grounded petrels

Over the four years we collected 59 Manx Shearwaters, 45 Leach's Storm-petrels and 1 European Storm-petrel (Table 1). Fewer than 3% of birds were found dead in this study, all in 2006 (Table 1). They included one Leach's Storm-petrel which had become trapped in an open drain-hole and drowned, another which had landed in an open and partially-full diesel sump and become entirely saturated in fuel, and, exceptionally, one Manx Shearwater found hanging next to an outside light with its head lodged in a ventilation grill and its neck broken. Subsequently the drain-hole was covered and the diesel sump kept drained and dry at all times. All other birds (>97%) were found alive, and successfully released to sea on the same day.

Between- and within-year differences in petrel numbers and artificial light

Numbers of Leach's Storm-petrels found in the village (Table 1) differed significantly between years

Table 1. Numbers of Leach's Storm-petrels *Oceanodroma leucorhoa*, European Storm-petrels *Hydrobates pelagicus* and Manx Shearwaters *Puffinus puffinus* found between 1 September and 16 October around buildings in the village on Hirta, St Kilda, in different conditions of artificial lighting, from 2005 to 2008. Counts given in parentheses are numbers of birds found dead.

		Number of birds found			
Year	Village night lighting	Leach's Storm-petrel	European Storm-petrel	Manx Shearwater	Total
2005	On	11	0	5	16
2006	On	28(2)	0	10(1)	38
2007	Reduced	0	0	27	27
2008	Reduced (1 Jul–21 Sept) (12 Oct–16 Oct)	0	0	0	0
	On (22 Sept–11 Oct)	6	1	17	24
Total		45	1	59	105

(test for homogeneity: $\chi^2_3 = 38.65$, $P < 0.01$). The only year that none were found was 2007, when village lighting was reduced for the entire autumn period. In 2008, numbers of Leach's Storm-petrels differed significantly between periods with and without light reduction measures in place (test for homogeneity: $\chi^2_1 = 4.16$, $P < 0.05$); however, the total number of individuals found was very small (Table 1). Birds were found only during the period when light reduction measures were not in use, and the first individuals were discovered on the morning of 23 September, immediately following the first night that outdoor lights were on and lighting left uncovered in the village (Fig. 1). Leach's Storm-petrels were never found during any time in this study when measures to minimize artificial light emissions were in place. Numbers of Manx Shearwaters found in the village (Table 1) also differed significantly between years (test for homogeneity: $\chi^2_3 = 18.48$, $P < 0.01$). Unlike Leach's Storm-petrels, Manx Shearwaters were found in all years, including 2007 (Table 1 & Fig. 1). In 2008, numbers of Manx Shearwaters differed between periods with and without light reduction measures implemented at night (test for homogeneity: $\chi^2_1 = 15.06$, $P < 0.01$). Manx Shearwaters were found only within the 20-night period that no light reduction measures were in place and the first on the morning of 23 September, immediately following lights first going on (Fig. 1). Also during this period, on 4 October, the only European Storm-petrel of the study was found.

Effects of the moon on petrels and shearwater responses to artificial light

The number of Leach's Storm-petrels and Manx Shearwaters found in the village attracted to lights was influenced significantly by the percentage of the moon's face illuminated at night (GLM: $z = -3.768$, $P < 0.001$) and by the percentage of the night that the moon was above the horizon (GLM: $z = -2.243$, $P < 0.05$), with no significant interactions. Species was tested in the model as an additional explanatory variable and effects found to be non-significant. Figure 1 shows that, overall, the vast majority of Leach's Storm-petrels and Manx Shearwaters were found at times of very low moonlight, for example after nights when less than 20% of the moon's face was illuminated and after nights when the moon was above the horizon at sea level for less than 20% of the time between sunset and sunrise.

DISCUSSION

Assessment of numbers and ages of petrels attracted to artificial lights

Total numbers of Leach's Storm-petrels, European Storm-petrels and Manx Shearwaters found during this study (Table 1) were very low compared with the estimated combined total of over 60 000 individuals of these species which have annually bred at St Kilda (Mitchell *et al.* 2004, Newson *et al.* 2008). Adult breeding activity of these species may continue at the colonies until November (Brooke 2004), but only juvenile petrels were found during the study period. This strongly suggests that, in September and October, adults are not normally influenced by artificial lighting at night on Hirta. Outside of this period, it is likely that effects of the lighting on adults are also minimal. Only Manx Shearwaters have ever been found attracted to lights on St Kilda outside of the species' normal fledging times. Fewer than ten have been reported in total, and all were thought to be early or late fledglings, based on the time of year (all broadly within the autumn period) and presence of chick down in their plumage (S. Murray pers. comm. 2009, *St Kilda Rangers' Reports* 1969–2008, Snow & Perrins 1998, Brooke 2004).

Juveniles were the only age group attracted by artificial lights on Hirta in this study, but it is difficult to state the scale of effects precisely. Measures of Leach's Storm-petrel, European Storm-petrel and Manx Shearwater productivity do not exist for all years of the study, so estimates of the proportions of the total number of fledged juveniles that were attracted to lights each year cannot be determined for all species. However, it is very likely that such estimates would be extremely small, as very low numbers of petrels were found in comparison to the most recent estimates of breeding population sizes at St Kilda (Mitchell *et al.* 2004, Newson *et al.* 2008).

Effects of artificial light reduction and moonlight

Between-year differences in numbers of Leach's Storm-petrels found in the village were probably because of deliberate reductions in light emissions rather than other unknown year effects. There was a significant within-year difference in numbers of Leach's Storm-petrels found in 2008, between the times when light reduction methods were in place and the deliberate control period when light emissions were not reduced.

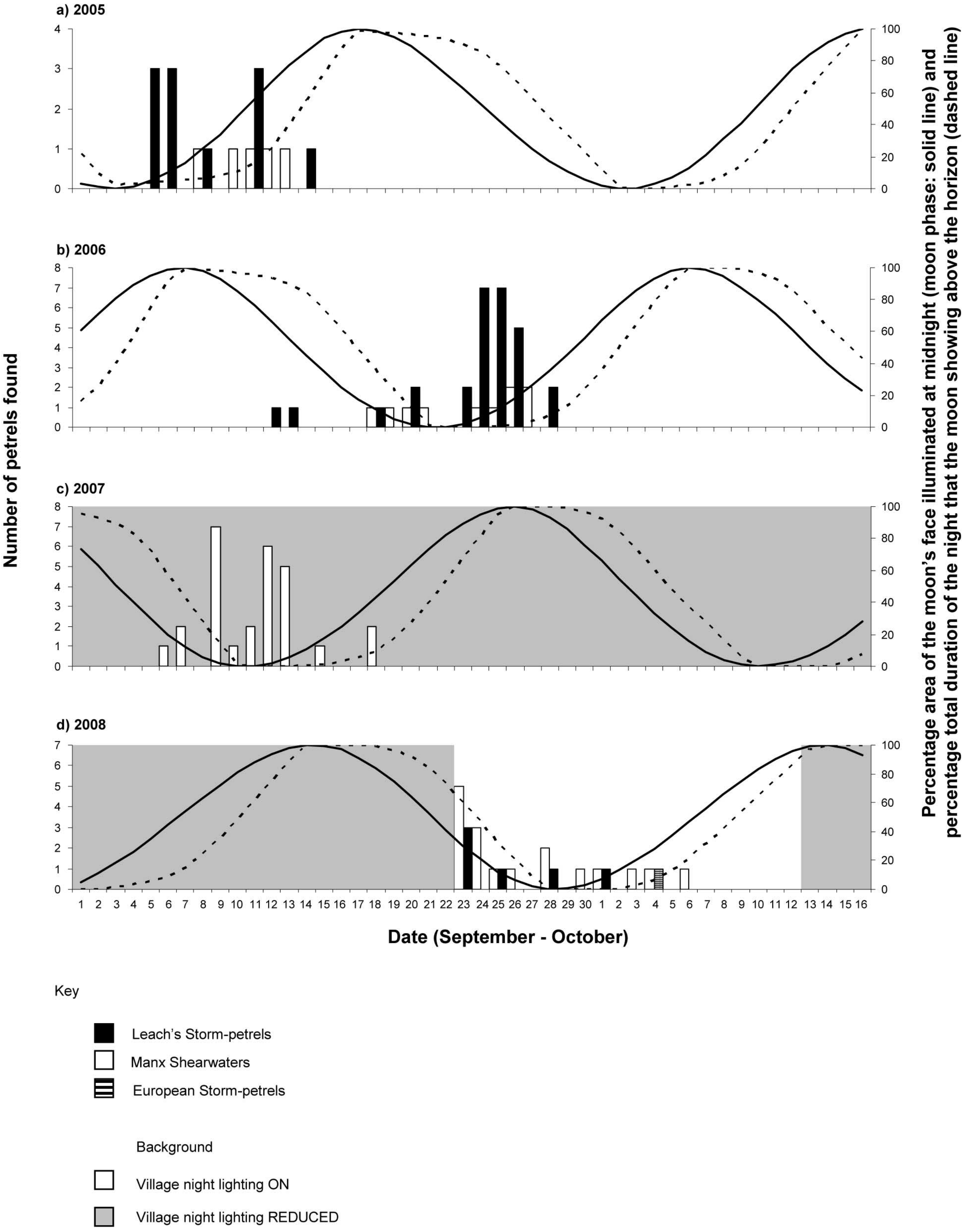

Figure 1. Distribution of numbers of Manx Shearwaters *Puffinus puffinus*, Leach's Storm-petrels *Oceanodroma leucorhoa* and European Storm-petrels *Hydrobates pelagicus* found around buildings in the village on Hirta, St Kilda, between 1 September and 16 October, with different conditions of artificial lighting (background), moon phase (solid line), and total duration that the moon was above the horizon at sea level at night (dashed line) from 2005 to 2008.

Also, numbers of Leach's Storm-petrels found in the village differed significantly between 2007, when lights were off, and the period in 2008 when lights were on (test for homogeneity: $\chi^2_1 = 4.16$, $P < 0.05$). These differences are very suggestive that differences in numbers of birds found in different years occurred in response to deliberate changes in artificial light conditions. Considering the two most obvious possible other influences (annual productivity and annual weather effects), there is little to suggest that differences in numbers were not due to the deliberate changes in light reduction measures. Productivity estimates for the species in 2007 and 2008 were not significantly different (Money *et al.* 2008, Money unpublished data) and although particularly high numbers of seabirds are found during foggy weather (Harrow 1976, Verheijen 1981, Warham 1990, Jones 2001), low-visibility and extreme weather conditions in the village on Hirta were recorded very infrequently in 2007 and 2008 (*St Kilda Ranger's Report* 2007, 2008). It was unfortunate that, owing to practical limitations, the start of the period with lighting on in 2008 could not be chosen entirely randomly, so experimental control was not perfect in this respect. However, the results showed no indication of being an artefact of experimental design and, considering this potential bias alongside the other three years' data, overall, the data seemed strongly indicative and convincing that deliberate reductions to light emissions during this study reduced attraction of Leach's Storm-petrels.

Unlike Leach's Storm-petrels, a high number of Manx Shearwaters was found in 2007, and effects of reducing light emissions on the numbers of birds attracted to the village were apparently not the same for Manx Shearwaters as for Leach's Storm-petrels that year. Given the measures in place to reduce lighting to the absolute minimum throughout 2007, it seems possible that Manx Shearwaters may still be attracted by very weak lighting, even the extremely low-level emissions on St Kilda in 2007 which did not affect the smaller species of petrel breeding at the site. Greater sensitivity to artificial lights in larger species of petrels has been suggested in other studies, for example differences between shearwaters and storm-petrels in Hawaii and in the Canary Islands (Telfer *et al.* 1987, Rodríguez & Rodríguez 2009). Additional evidence for this theory at St Kilda is that European Storm-petrels are the smallest species to breed on Hirta and nest in walls next to the MoD base (unlike Leach's Storm-petrels and Manx Shearwaters), yet have hardly ever been found grounded around buildings at any time of year, and are apparently the least sensitive to light (*St Kilda Rangers' Reports* 1969–2008, Murray 2002, Miles & Money pers. obs.). One other possibility, however, is that Manx Shearwaters may be more attracted by sounds at night than storm-petrels and that certain noises continue to attract shearwaters at times when artificial lighting is minimal or even non-existent. On Hirta, Manx Shearwaters have most frequently been found close to extractors and generators that were continuously emitting low frequency sounds, including in all years of this study. It seems likely that attraction to these sound emissions could be one explanation as to why Manx Shearwaters were found in 2007 during reduced light conditions.

Effects of the lunar cycle and position of the moon above the horizon on numbers of grounded petrels were similar in this study to those found in other studies: most petrels were found at times of least moonlight (Verheijen 1980, Telfer *et al.* 1987, Le Corre *et al.* 2002, Rodríguez & Rodríguez 2009). In 2008, a separate study was carried out on the phenology of Leach's Storm-petrels at St Kilda, in which fledging dates were recorded (Money unpublished data). Thirteen birds, out of 28 studied, fledged between the first and last quarter of the lunar cycle (7–22 September), including three on nights around the full moon (14–16 September). In other studies on light attraction of petrels, the possibility has been suggested that fewer juveniles have been found at times of greatest moonlight (e.g. full moon) because fledging was inhibited on these nights (Imber 1975, Rodríguez & Rodríguez 2009). However, for Leach's Storm-petrels at St Kilda in 2008, the phenology study suggested this was not so. The lack of grounded Leach's Storm-petrels at times of greatest moonlight was perhaps more probably because of the relative glare and attraction of artificial lights diminishing on nights when ambient light from the moon was particularly bright and long-lasting.

Occurrences of light-induced mortality of petrels at St Kilda and in the UK

Mortality of petrels found attracted to lights was very low at St Kilda (<3%). This has also been found in similar studies on much larger and more populated islands (more petrels and more people), for example Réunion Island (<10%) and Tenerife (<6%) (Le Corre *et al.* 2002, Rodríguez & Rodríguez 2009). Considering the decline in Leach's Storm-petrels reported from Dùn (Newson *et al.* 2008), in relation to our results, the possibility that high mortality of breeding and non-breeding storm-petrels may have occurred in the UK away from St Kilda was reviewed, by searching all regional bird reports and county avifaunas for records of light-induced effects and mortality of Leach's and European Storm-petrels, for all areas of the UK with storm-petrel breeding colonies (Mitchell *et al.* 2004), in all years from 1990 to 2006 (Table 2). Most frequent were records of attraction to lighthouses and harbour lighting, but unusual records included: individuals coming to flashes from a garden fireworks display (Egilsay, Orkney, 5 November 2005); attraction to oil terminal flares (Sullom Voe, Shetland, 3 November 2000); and several individuals attracted to moth traps (Skaw, Shetland, 30 July 2004). Given the time period and area covered (Shetland, Orkney, all regions of the UK north and west coasts, Scillies, and the Channel Islands), records were surprisingly few in total (<120 individuals). However, the proportion of all records of storm-petrels found dead (21%) was high in comparison with our study at St Kilda (<2%). Perhaps because records from regions of the UK other than St Kilda were not all made systematically, they were possibly biased by a greater likelihood of dead birds being found during casual observations than live and potentially transitory individuals. Even with this consideration, the results of this search strongly suggest that in areas of the UK with storm-petrel breeding colonies away from St Kilda, mortality of Leach's Storm-petrels and European Storm-petrels owing to light attraction has also been very low in comparison to estimated UK breeding population sizes (Mitchell *et al.* 2004). It was notable that the highest proportions of all Leach's Storm-petrel and European Storm-petrel records (70% and 86.6%, respectively) came from Bardsey lighthouse. This may partly be explained by relatively high observer coverage at this light source, but even taking this into account, this site has a high attraction power to birds in comparison with other intensively watched sites with lighthouses, such as North Ronaldsay and Fair Isle (Bardsey, Fair Isle, North Ronaldsay, and Orkney *Bird Reports* 1990–2006). Possible reasons suggested for this have included differences in lighthouse beam characteristics (e.g. light frequency and rotation rate), as well as site location differences relative to species' migration routes and breeding areas, migration bottlenecks, seabird foraging ranges, and seasonal and local weather patterns (Saunders 1930, Herbert 1970, Verheijen 1981, *BardseyBird Reports* 1990–2006, Brooke 1990, Jones 2001, Jones & Francis 2003, D. Shaw pers. comm. 2009). Overall, in areas of the UK with breeding storm-petrels, it seems that very low numbers of individuals are affected by artificial lighting relative to estimates of total breeding population sizes, and light attraction is not a cause of high mortality.

Table 2. Total annual numbers and mortality of Leach's Storm-petrels *Oceanodroma leucorhoa* and European Storm-petrels *Hydrobates pelagicus* recorded at artificial light sources in regional bird reports and avifaunas for all regions of the UK with storm-petrel breeding colonies other than St Kilda, for all years from 1990 to 2006.

Year	Leach's Storm-petrels found	European Storm-petrels found
1990	2	8
1991	0	2
1992	1	2
1993	0	2
1994	3	8
1995	2	3
1996	0	2
1997	4	0
1998	2	2
1999	1	6
2000	3	3
2001	0	0
2002	8	16
2003	0	1
2004	3	16
2005	1	9
2006	0	2
Total	30	82
Total found dead	8 (26.7%)	15 (18.3%)

The status of St Kilda as a Site of Special Scientific Interest, Special Protection Area, and World Heritage site means that increases in the number of brightly-lit buildings on the archipelago are unlikely. The inhabited village is the only area of Hirta with lighting on at night, with the exception of one MoD building on the hilltop, which has outside lighting occasionally left on. Petrels have been discovered near this building during the daytime by staff of the radar base, but very few birds have been found, less than annually, and the vast majority of these were alive. There are no other sources of artificial light on land at St Kilda and it is rare for brightly-lit ships to anchor for long near the islands. In conclusion, numbers of petrels attracted to artificial lights on St Kilda are low, very few are killed by the phenomenon, but reductions to artificial light emissions should be encouraged since they are beneficial in reducing numbers of fledglings that are grounded.

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A paradigm for understanding whole ecosystem effects of offshore wind farms in shelf seas

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Abstract

With the rapid expansion of offshore windfarms (OWFs) globally, there is an urgent need to assess and predict effects on marine species, habitats, and ecosystem functioning. Doing so at shelf-wide scale while simultaneously accounting for the concurrent influence of climate change will require dynamic, multitrophic, multiscalar, ecosystem-centric approaches. However, as such studies and the study system itself (shelf seas) are complex, we propose to structure future environmental research according to the investigative cycle framework. This will allow the formulation and testing of specific hypotheses built on ecological theory, thereby streamlining the process, and allowing adaptability in the face of technological advancements (e.g. floating offshore wind) and shifting socio-economic and political climates. We outline a strategy by which to accelerate our understanding of environmental effects of OWF development on shelf seas, which is illustrated throughout by a North Sea case study. Priorities for future studies include ascertaining the extent to which OWFs may change levels of primary production; whether wind energy extraction will have knock-on effects on biophysical ecosystem drivers; whether pelagic fishes mediate changes in top predator distributions over space and time; and how any effects observed at localized levels will scale and interact with climate change and fisheries displacement effects.

Keywords: marine renewable energy; bio-physical indicators; predator–prey interactions; scaling; multitrophic; autonomous platforms; dynamic Bayesian network modelling; cumulative impact assessment

Background: the need for (structured) speed

The timeframe with which to achieve climate-resilient development is rapidly narrowing (IPCC 2023). Therefore, governing bodies have increasingly turned to renewable energy technologies to try and meet electricity demands sustainably. In the marine space, this has resulted in the unprecedented growth of the offshore wind sector. Global offshore capacity is forecasted to increase by a factor of 10 by 2030 to reach 330 gigawatts (GW); this amount is projected to treble again by 2050 (IRENA 2019). Recent progress in the development of floating offshore wind technology, allowing turbines to be deployed in deeper waters, as well as hybridization with existing offshore infrastructure (e.g. using wind energy to power oil and gas installations) is also contributing to this burgeoning industry (Leporini et al. 2019). Consequently, more offshore windfarms (OWFs) are projected to occupy larger areas of midlatitude coastal and shelf-sea space (Díaz and Guedes Soares 2020). While OWFs are widely acknowledged to be part of the climate resilient development portfolio, any such resiliency will depend on the ability to safeguard biodiversity and ecosystem functioning, especially salient given the concurrent pressures of ocean warming, acidification, and deoxygenation (Williamson and Guinder 2021). Therefore, as we speed up offshore wind development of shelf seas, so too do we need to accelerate our understanding and ability to predict their effects on shelf sea ecosystems.

It is also important to be able to distinguish between the different potential causes of any measured and predicted effects, including those from OWFs, climate change, and the natural variability of shelf seas over multiple spatial and temporal scales (Trifonova et al. 2022a). Achieving this at the rapid pace of industry development will necessitate a shift towards more explicit ecosystem-wide and cumulative effects theory and implementation (Trifonova et al. 2022b, Declerck et al. 2023). This will require prioritizing hypothesis-driven study of ecosystem processes, as well as individual species' populations and the trophic level interactions between populations at spatial and temporal scales relevant to all these processes and OWFs. Performing such science has been limited so far due to the constraints, chiefly economic, logistic, and technological, that come with studying the sea (Godø et al. 2014a). Fortunately, in the past decade, there have been huge

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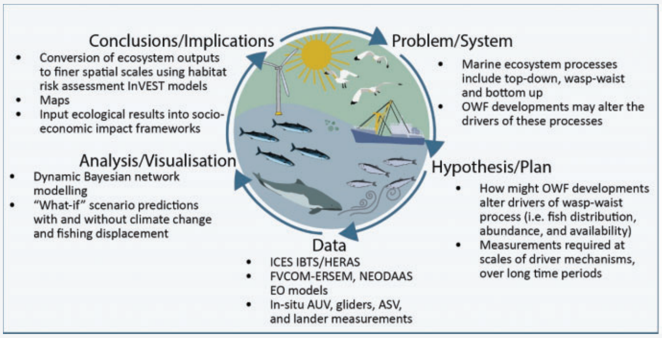
Box 1. Fish as the fulcrum

Problem/Background: OWFs are increasing in number in the North Sea, while simultaneously our understanding of this sea remains limited. However, in temperate shelf seas, just a few pelagic fish species play a pivotal role at intermediate trophic levels by exerting top-down control on zooplankton and bottom-up control on top predators ('wasp-waist' processes). These species occupy multiple habitats throughout their life cycle and have migratory routes between spawning and nursery habitats which are influenced by a range of bio-physical drivers (e.g., currents, temperature, levels of stratification, tidal mixing fronts and plankton production). The abundance, distribution and growth rates of these fishes, therefore, provide the link for regulating trophic dynamics between primary producers and top-predators. Should OWF developments alter, either indirectly or directly, any of the drivers of the fish life cycle, then any resultant changes to fish are likely to propagate across shelf-sea food webs.

Hypothesis/Plan: By measuring effects of OWFs on fish, it will be possible to, with necessary levels of certainty, predict effects of OWF on lower and higher trophic levels simultaneously, for the entirety of the North Sea. To do so will require identifying and characterizing the drivers of key pelagic fish species' abundance, availability (to top predators), distribution and growth rates. Measurements will need to be taken at the scales of driver mechanisms, throughout the year, thereby capturing the full life cycle of fish and the habitats they influence and are influenced by.



Data: Historical data on fish, seabird and marine mammal distributions will be used to identify optimal sampling areas. The ICES IBTS and HERAS surveys as well as commercial catch data can provide data on herring, sandeel, sprat and mackerel. NEODAAS Earth Observation data synthesized from satellites on ocean temperature and colour from products such as chlorophyll-a, can provided output on locations of fronts.

Bio-physical coupled oceanographic models (FVCOM + ERSEM) can predict changes in levels of primary production, changes in stratification and hydrodynamic regimes. Mechanistic changes in the subsurface pelagic environment will be measured *in situ* using AUVs, gliders, ASVs, and landers equipped with hydroacoustic and oceanographic sensors (e.g., multifrequency echosounder, ADCP, CTD, fluorometer). These will be strategically deployed in areas previously identified from historical data during periods of contrasting biophysical regimes in the North Sea (spring, summer and autumn).

Analysis/Visualisation: Dynamic Bayesian network (ecosystem) models (DBN) capable of resolving spatio-temporal patterns across multiple trophic levels will be used to predict the main drivers of species population change. Integrated Nested Laplace Approximation (INLA) models will use the identified drivers and run high spatial resolution analysis to explain variation in fish migration routes, growth rates, and links with top predator distributions.

A range of "what-if" scenario predictions with and without climate change and fisheries displacement effects will be modelled with the DBN and compared to the baseline runs to be able to separate these co-occurring yet potentially obfuscating effects from those of OWFs.

Conclusions/Implications: Outputs from the ecological analysis will be fed into the development of holistic risk-benefit maps using the InVEST model. The evidence generated will also influence the generation of strategic frameworks to structure and improve Cumulative Impact Assessments; the scientific evidence-base will thereby facilitate transparent communications between academia, developers, decision-makers, and the wider public. Finally, any insights gleaned from the process as well as the results themselves will inform the generation and testing of further hypotheses. As technologies, and the socio-economic and geo-political climate evolves, these will form the basis of a new iteration of the investigative cycle.

advancements in sensors (remote as well as *in situ*) and autonomous platforms that can concurrently measure ecosystem processes, but also in statistical modelling approaches and the computing power required to run them. The resultant temptation towards abundant data collection resulting in 'data-rich but information-poor' (DRIP) syndrome (Wilding et al. 2017), must and can be avoided by formulating concrete hypotheses rooted in ecological theory.

Adapting established frameworks for conducting investigative science is a useful starting point, one widely recognized framework within this cycle is the PPDAC model, which stands for Problem, Plan, Data, Analysis, and Conclusions (Tukey 1980, Mackay and Oldford 1994, Wild and Pfannkuch 1999) (Fig. 1). The cycle begins by identifying and defining the problem (hypothesis generation), after which, experiments are designed, data are collected, analysed, and finally, interpreted. The conclusions reached in the final stage then become the basis for the next iteration of the cycle. In the face of urgency, PPDAC provides a systematic framework, ensuring a thorough understanding of the problem before developing a well-considered plan. The iterative nature allows for flexibility in adjusting strategies based on evolving data and insights, crucial for addressing the dynamic and interconnected nature of complex issues. Notably, the statistics and data science community advocate for PPDAC as it improves data literacy in an age of 'big data' (Gehrke et al. 2021). The following sections detail how applying such a framework to the specific context of expanding OWF development can accelerate our understanding of and ability to predict effects on shelf-sea ecosystems. A case study exemplifying the cycle in the context of OWF effects on wasp-waist processes in the North Sea is also provided, to facilitate similar necessary research (Box 1).

The problem: the seascape of OWF effects

Numerous reviews of the potential environmental effects of OWFs in various seas have identified a multitude of concerns, including biodiversity loss, nonindigenous species propagation, fishery resource management, increases/decreases in primary production and chemical and noise pollution, degradation/enhancement of the seafloor, and alterations in food webs and ocean hydrodynamics (e.g. Galparsoro et al. 2022). Positive effects such as functional habitat increase for bentho–pelagic organisms have also been described (Gill et al. 2020). Further information is now readily retrievable via the Offshore Wind Farm Environmental Evidence Database (https://ories.pml.space). However, the bulk of evidence reviewed comes from studies conducted at individual turbine or single windfarm level, often on a particular species or community of interest (i.e. visible top predators).

The expansion of large-scale OWF into deeper waters has the potential to alter local and regional shelf-sea hydrodynamics and subsequently bio-physical processes, particularly in seasonally stratified areas that play a vital role in regulating prey availability for higher trophic levels (Dorrell et al. 2022). Changes to water currents, wind wakes (i.e. reduced kinetic energy on the leeward side of OWF), and turbulence induced by OWF may modify vertical stratification, changing mixed layer depths immediately 'downwind' of developments as well as vertical mixing levels over large spatial scales (Carpenter et al. 2016, Gill et al. 2020, van Berkel et al. 2020). This may have subsequent effects on the stability and strength of oceanographic features such as tidal mixing fronts (Simpson and Sharples 2012), which occur at the interface between well-mixed and stratified waters, as well as finer-scale internal waves occurring on the depth(s) of the edges of offshore banks that are localized sources of new primary production

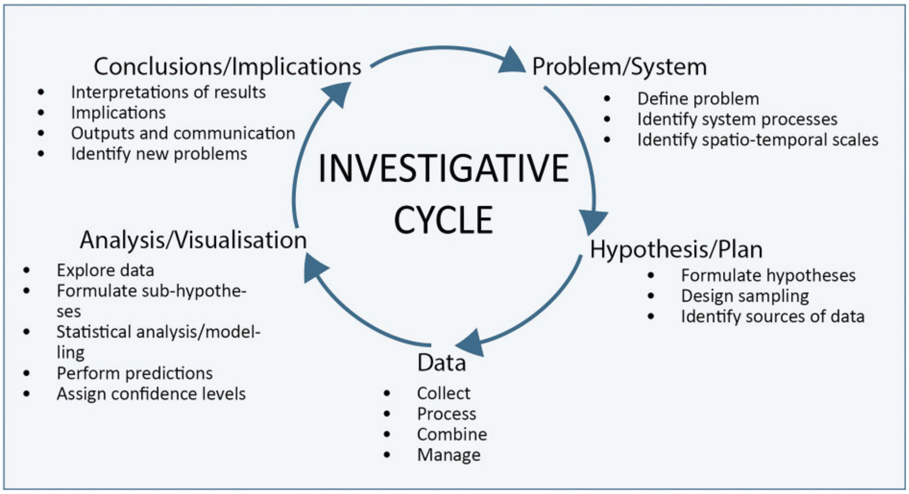


Figure 1. Visualization of the scientific investigative cycle, adapted from the PPDAC cycle proposed by Wild and Pfannkuch (1999).

(Palmer et al. 2008). These, in turn, have the potential to affect the timing and strength of diapycnal mixing, which subsequently controls the supply of nutrients in surface waters to create temporally persistent and predictable foraging areas for pelagic fishes and top predators (Cazenave et al. 2016). Changes in hydrodynamic regimes (i.e. levels of mixing, surface wave energy, and upwelling) could thus affect the nutrient supply to the euphotic layer and change its spatial pattern, with important knock-on effects for primary and secondary production (Floeter et al. 2017).

Therefore, while valuable, the focus on documenting distributional changes of few species does not necessarily scale to predictions about ecosystem-wide effects in shelf seas, an inherently complex system (Steele et al. 1989). This, in turn, hampers the ability to attribute a cause-and-effect relationship between offshore wind developments and receptor populations of interest; this also contributes to critical levels of uncertainty in cumulative effects studies (Goodale and Milman 2016). Furthermore, it cannot be assumed that the summation of changes in cumulative effects studies to individual species under worst-case scenarios accurately predicts outcomes at population levels, let alone ecosystem or regional scales (Nogues et al. 2023). To be able to do so requires studying and understanding the physical and biological processes underlying the changes in species distributions and abundances, from bottom-up drivers to top-down pressure as well as trophic interactions (Levin 1992). Put simply, we need to study the processes that drive distributions at the spatial and temporal scales at which they occur. Only then will the environmental evidence base be able to support holistic cumulative effects assessments at the large scales and at the rapid pace proposed for the offshore wind industry (Box 1).

The plan: ecosystem processes and indicators of change in space and time

Ecosystems in temperate midlatitude shelf seas are characterized by three processes: (1) bottom-up processes that regulate primary production dynamics (Simpson and Sharples 2012); (2) top-down processes arising from predation and anthropogenic pressures (e.g. via commercial fishing) (Lynam and Mackinson 2015); and (3) wasp-waist processes regulated by a few midtrophic pelagic fish species, which provide the critical fulcrum in linking (1) and (2) (Cury et al. 2000) (Fig. 2). Testing these ecological theories in the context of offshore wind effects is therefore merited; an expedient approach is to focus on studying wasp-waist processes as they mediate both lower and higher trophic levels (Box 1).

Due to the dynamic and mobile nature of the marine environment and its wildlife, indicators and mechanisms of the three ecosystem processes (bottom-up, top-down, and wasp-waist) operate at distinct yet interconnected ranges in space and time. These scales at which the processes and main species within trophic levels occur and interact at are almost linear: primary level (<1–10 km, hours to days); secondary level (10–100s km, days to months); and tertiary level (100s–1000s km, months to years). Therefore, any studies aimed at assessing shelf-sea-wide effects will need to take these relevant scales into account (Pittman et al. 2021). The following section suggests definitions of spatial and temporal scales (adapted from Trifonova et al. 2022a) relevant to OWF and temperate midlatitude shelf sea systems and expands on important bio-physical features and indicators and their corresponding scales (Fig. 3).

Space

The fine spatial scale (<1 km) provides a mechanistic understanding of processes underlying both the production of plankton (primary trophic level) and the foraging behaviour of individual prey and predators (i.e. fishes, marine mammals, seabirds), including conditions for successful foraging connected to local (temporally varying) hydrodynamic conditions. Physical and topographic controls on local primary production, such as shelf edges, tidal mixing fronts, and internal waves are important at this scale, especially highly predictable productive areas, such as the edges of banks where internal waves enhance the aggregation of prey (Embling et al. 2013), which can influence top-predator distributions (Scales et al. 2014). These features are temporally ephemeral but predictable, as they are driven by daily (ebb-flood) as well as biweekly (neap-spring) tidal cycles and seasonal solar and wind field changes (Simpson and Sharples 2012). Competing effects of local scouring and changes in downstream mixing also alter how sediment is resuspended from the seabed, with

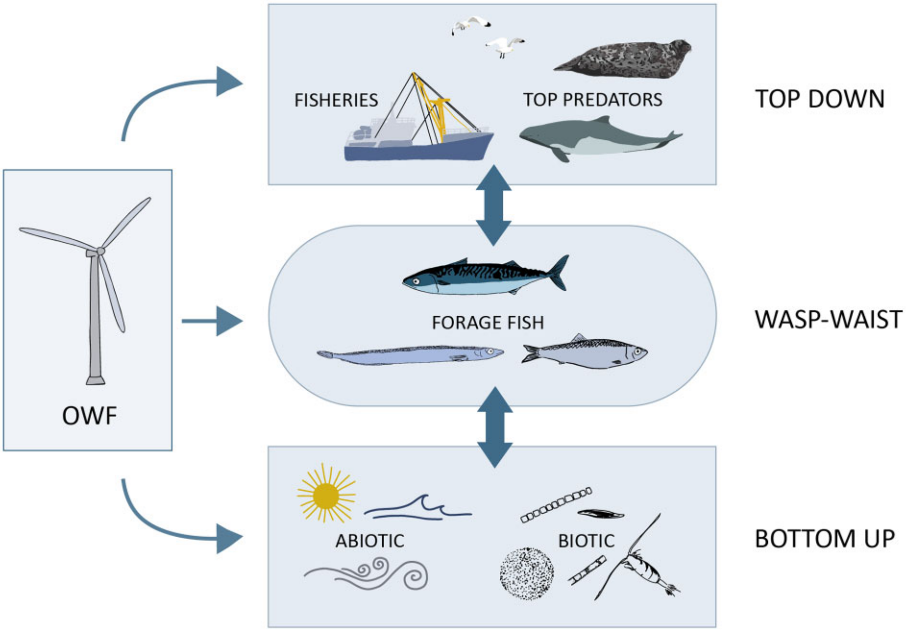
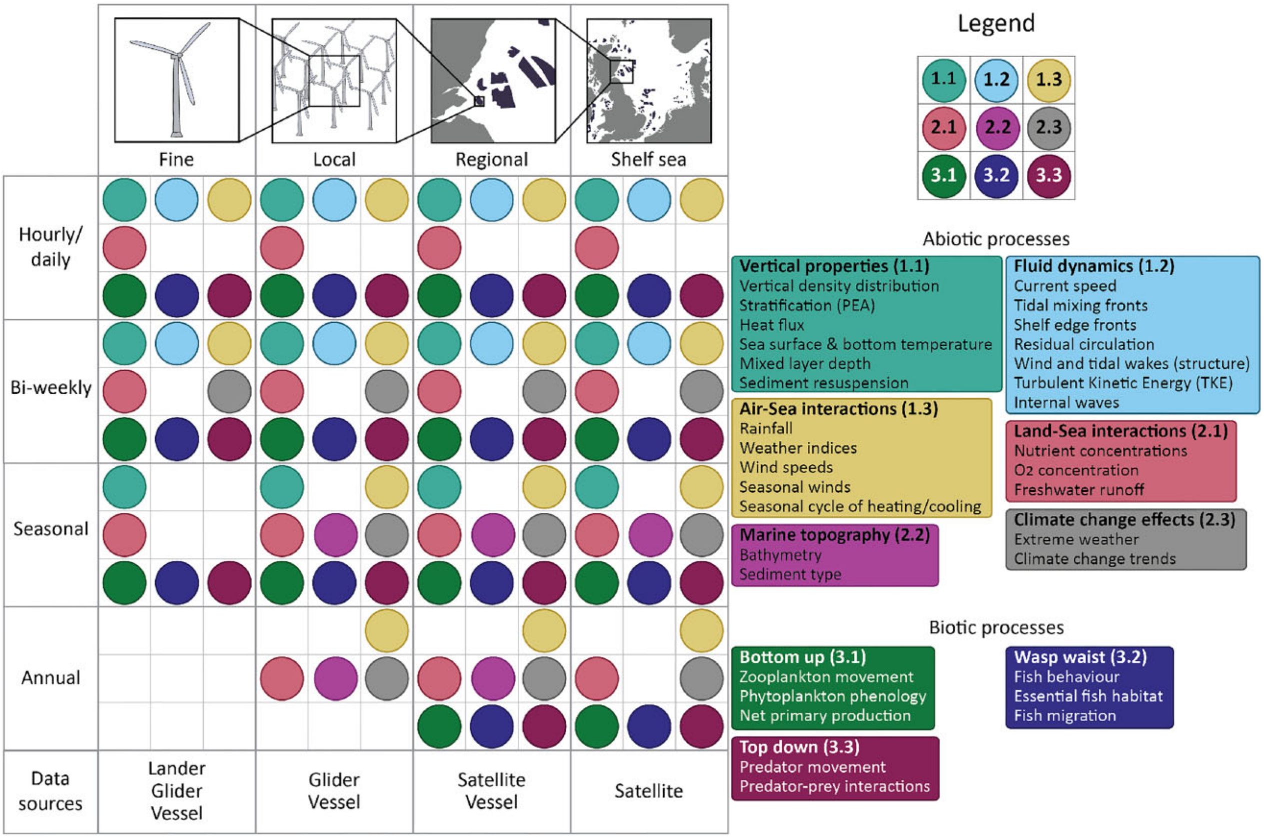


Figure 2. Pathways of OWF effects on top-down, wasp-waist, and bottom-up environmental processes in shelf seas.



	Fine	Local	Regional	Shelf sea
Hourly/ daily	1.1 1.2 1.3 2.1 3.1 3.2 3.3	1.1 1.2 1.3 2.1 3.1 3.2 3.3	1.1 1.2 1.3 2.1 3.1 3.2 3.3	1.1 1.2 1.3 2.1 3.1 3.2 3.3
Bi-weekly	1.1 1.2 1.3 2.1 2.3 3.1 3.2 3.3	1.1 1.2 1.3 2.1 2.3 3.1 3.2 3.3	1.1 1.2 1.3 2.1 2.3 3.1 3.2 3.3	1.1 1.2 1.3 2.1 2.3 3.1 3.2 3.3
Seasonal	1.1 2.1 3.1 3.2 3.3	1.1 1.3 2.1 2.2 2.3 3.1 3.2 3.3	1.1 1.3 2.1 2.2 2.3 3.1 3.2 3.3	1.1 1.3 2.1 2.2 2.3 3.1 3.2 3.3
Annual		1.3 2.1 2.2 2.3	1.3 2.1 2.2 2.3 3.1 3.2 3.3	1.3 2.1 2.3 3.1 3.2 3.3
Data sources	Lander Glider Vessel	Glider Vessel	Satellite Vessel	Satellite

Figure 3. Shelf sea biophysical processes and the principal space and time scales at which they operate, including recommendations for how experimental data can be collected; each circle in the table represents the full range of the process in question, although specific subprocesses (i.e. zooplankton movement) may operate within fewer grid squares. PEA = Potential Energy Anomaly.

potential impacts on fluxes of greenhouse gases, nutrients, and light penetration, therefore, all effecting primary production with knock-on effects on 'blue carbon' storage (Dorrell et al. 2022). At the very fine scale (i.e. changes in turbulence from the subsurface mixing around pylons or jackets and the wind wake of turbine structures from wind energy extraction), interactions between currents and turbines may produce physical features that influence top predators targeting of locations due to changes in the level of turbulence in the water column (Lieber et al. 2019, Schultze et al. 2020). How these physical processes develop at the scale of the wind farm array and for multiple arrays, and how they might affect top predator foraging [e.g. by structures acting as fish aggregation devices (van Berkel et al. 2020), and/or by locally changing levels of primary production (Slavik et al. 2019)], require further study.

Net primary productivity and the depth and biomass concentration of the subsurface chlorophyll-a maximum (SCM) are relevant indicators at local spatial scales (1–100 km), as are variables linked to physical aspects such as seabed topography, stratification, and weather (e.g. rainfall, wind speed, heat exchange) (Holt et al. 2012, Sharples et al. 2013a). The survival and community structure of phytoplankton are dependent upon the marginally stable pycnocline being maintained by weak levels of diapycnal mixing that injects nutrients upward (Palmer et al. 2008). Small changes may have dramatic but nonlinear effects on physical water column structure, the timing, and magnitude of primary productivity, and thereby primary ecosystem function. Such effects may be monitored as changes in dissolved oxygen concentration, a key indicator of ocean health, which is regulated by both physical mixing and biological production and consumption rates (Williams et al. 2022). Fine-scale features and mechanisms are likely to inform on bottom-up regulations of marine ecosystem functioning and are likely to predict the distributions of fish, top predators, and fisheries (Trifonova et al. 2017).

The regional domain (100s–1000s of km) is characterized by populations and meta-populations of large mobile species (seabirds and marine mammals), where seasonal mean stratification, bottom temperature (BT), net primary production, and maximum chlorophyll-a values appear as principal indicators of their density distribution (Scott et al. 2010, Cox et al. 2018). Stratification and BT are also important for the abundance, distribution, and diversity of many fish species (Sagarese et al. 2014).

Finally, the largest scale extends over most of the world's shallow shelf seas (1000s of km, e.g. the North Sea, China Sea, etc.), where interconnectivity between regions is provided by the migrations of large mobile nekton (i.e. fish, seabirds, marine mammals) (Hammond et al. 2013, Rutterford et al. 2015). Net primary production, mean BT, and maximum chlorophyll-a are still important indicators at this scale, where the timing of seasonal phytoplankton blooms is an extremely important indicator for marine food web functioning and energy flow (Friedland et al. 2018, Silva et al. 2021). Predator–prey interactions at this scale and how they vary on a seasonal and inter-annual basis is also vital, as changes in migration patterns of pelagic fish species may have important knock-on effects for higher trophic levels (Samarra and Foote 2015). Top-down control via fisheries exclusion by OWFs may also result in population-level effects; the dynamics between fisheries, stocks, and offshore wind will be increasingly relevant to ascertain as floating wind technology allowing for expansion of wind developments into deeper waters progresses (Gill et al. 2020, Farr et al. 2021).

Time

There are various temporal scales to consider where there are processes that are linearly linked to the increasing spatial scales discussed above. The daily ebb-flood tidal cycle (ca. 12.5 hours) and diurnal cycle (24 hours) are the smaller timescales during which behaviours such as the diel vertical migration (DVM) of fish and zooplankton through the water column have consequences for energy transfer and interactions between predator and prey, and therefore, energy transfer across trophic levels (Castellani et al. 2013, Brierley 2014). The influence of tides (both the flood/ebb and spring/neap cycles) is especially important at local and fine spatial scales generally nearer the coast where tidally driven horizontal current speed and physical features both attract and make prey more available to top predators (Zamon et al. 2003, Cox et al. 2013).

At the seasonal scale of months, the annual timing of the spring phytoplankton bloom affects the survival of larval and juvenile fish (Platt et al. 2003), with consequences for higher trophic levels, as it has also been found to affect the breeding success of seabirds (Scott et al. 2006). This raises the question of whether trophic mismatches due to discrepancies between the timing of available prey and the start of breeding seasons will have implications for top predator population viability (Howells et al. 2017). Since phytoplankton blooms are dependent on the timing of stratification during spring in temperate continental shelves (Sharples et al. 2006), changes to the timing and strength of seasonal stratification of the water column will also affect critical links such as nutrient fluxes between benthic and pelagic habitats (Nunnally 2019).

On longer time scales (seasonal and annual), abiotic factors such as salinity, oxygen levels, and BT as well as biotic factors such as food availability has been found to affect the distribution, abundance, and species richness of fishes at a regional and shelf-wide scales (Sagarese et al. 2014, Merillet et al. 2020). Interannual variability in the timing and duration of the spring phytoplankton bloom and then subsequent summer season interactions with the spring–neap tidal cycle, and weather conditions, control the intensity of subsurface blooms ('biweekly blooms'). These are likely to affect fish recruitment and survivability, and through this influence fisheries production and (the predictability of) top predator distributions (Box 1). Notably, climate change is expected to disrupt the existing phenology between fish larvae and availability of zooplankton due to changes in timing of spring blooms (Dulvy et al. 2008). At the annual scale, some commercially important pelagic fish species (e.g. Atlantic herring *Clupea harengus*) have highly predictable annual migration routes to and from spawning and feeding grounds that help long-lived mobile predators learn when and where they will be available (Roff 1988). Predictable species-specific variation in annual and seasonal locations of fish migration routes and their links to frontal and surface primary production is likely to drive variation in top-predator (seabird and mammal) distributions (Warwick-Evans et al. 2016).

The data: fisheries and Earth observation datasets and advances in *in-situ* measurement techniques

Several existing fishery independent and dependent datasets are spatially explicit time-series that allow for the dynamic changes of fish distributions and abundances over large spatio-temporal scales to be quantified. These range from internationally coordinated fisheries independent scientific campaigns such as the North Sea International Council for the Exploration of the Sea (ICES) International Bottom Trawl Surveys (IBTS) to acoustic surveys for pelagic fish species such as the Herring Acoustic Survey (HERAS). Fishery-dependant data such as monthly commercial landings are generally available only as an aggregate product on a larger scale (i.e. an ICES square 30 × 30 nmi) and therefore represent less explicit locations of catch. However, these are still a valuable alternative or addition for examining temporally changing fish distributions in regions where fishery-independent surveys are unavailable (e.g. Marine Management Organization commercial landing by Exclusive Economic Zone; Dixon et al. 2019) Complementing these datasets with satellite-based vessel monitoring system (VMS) data (e.g. Alemany et al. 2014) and automatic identification system (AIS) data can also offer insights on fishing activity distribution and therefore top-down pressure in the ecosystem (Thoya et al. 2021).

Combining such fish datasets with remotely sensed Earth observation (EO) oceanographic data is especially conducive to the study of bottom-up processes. EO data provide continuous information of submesoscale (<10 km) through to microscale (<100 m) details of the global ocean's colour, structure, and circulation including discrete oceanographic features such as fronts and eddies (Belkin 2021). Therefore, the spatial and temporal scales over which these features persist and vary can be tracked, although this is only possible where distinct surface signatures are present (Miller 2009, Cox et al. 2018). While sea surface temperature (SST) and chlorophyll-a concentrations (colour fronts) derived from satellites are widely used to identify productivity hotspots for pelagic species, the continuous features of oceanographic fronts are typically not explicitly extracted despite their ecological significance (Belkin 2021). Studies combining frontal datasets with individual biotelemetry data have revealed the tendency of large fish species to track fronts and associated features during migration (Luo et al. 2015, Miller et al. 2015a); such approaches should increasingly be used for temperate pelagic fish species (Spondylidis et al. 2023). At much smaller spatial and temporal scales (i.e. <1 km, <1 week), the timing and location of surface concentrations and strong gradients can be derived from higher-resolution (300 m) ocean colour data [e.g. ESA's Medium Resolution Imaging Spectrometer (MERIS)]. The combined use of thermal and colour frontal distributions provides a more complete analysis of persistent biological and physical processes in shelf seas (Miller et al. 2015b). This can further provide a more holistic picture of pelagic frontal dynamics to facilitate the planning of future offshore renewable developments (Medina-Lopez et al. 2021).

To be able to elucidate the fine-scale mechanisms of drivers in marine ecosystems and capture any (predictable) variation, it is also necessary to study them in place and at the scales at which these operate. Advances in *in-situ* measurement techniques over the past decade now make it possible to study environmental drivers of ecosystem processes at these resolutions (Supplementary Tables S1 and S2). These include well-established active and passive acoustic techniques that allow for measurement of the spatio-temporal distribution and abundance of organisms as well as physical structuring such as the mixed layer depth and internal waves to be characterized (reviewed in Horne 2000, Godø et al. 2014a). Deploying echosounders (active) and hydrophones (passive) in tandem maximizes species discrimination capability and allows for the movements of organisms to be tracked (Williamson et al. 2021, Gillespie et al. 2022). Combining acoustic sensors with concurrent environmental measurements allows for multitrophic monitoring; this approach deployed on static landers is already well documented in the marine renewable energy industry (see e.g. Williamson et al. 2016, Cotter et al. 2017). As the offshore wind industry increasingly moves towards floating wind technology in deeper waters, the development of similar multisensor floating platforms and subsequent combination/integration with turbine structures and observation systems will become increasingly relevant.

Recent advances in uncrewed and autonomous vehicle technologies also offer huge potential to deliver a more complete understanding of shelf-sea ecosystems, as they allow for concurrent measurements of multiple trophic levels over large distances and durations at high spatial resolutions (Ludvigsen and Sørensen 2016). For instance, novel wind and wave-powered gliders that are acoustically silent as well as low fossil fuel emitting are now capable of long duration missions with both active and passive acoustic sensing (Verfuss et al. 2019). Attachment of oceanographic sensors (e.g. CTDs, chlorophyll and backscatter sensor, nutrient sensors, eDNA samplers) to gliders and Autonomous Underwater Vehicles (AUVs) is also recently made possible (Palmer et al. 2021). When deployed on AUVs, such data can be sent in near real-time, and thus assimilated into high resolution 3D oceanographic forecasting models (Cossarini et al. 2019). The attachment of eDNA samplers to AUVs in particular promises to open up new possibilities for biological monitoring as this technique allows for simultaneous multitrophic measurements, including occurrence of microbes, plankton, invertebrates, and fish (Yamahara et al. 2019). The development of new Photosynthetically Active Radiation (PAR) sensors combined with chlorophyll fluorescence measurements will also allow for rates of primary production to be calculated *in situ* (Loveday et al. 2022). As measuring rates of primary production is normally a time-consuming (and ship-based) process, measurements are sparse; this new *in-situ* method therefore has great potential to deliver data with better spatial coverage that progress understanding.

Returning to the bigger picture, it is the combination of measurements from fisheries trawls/surveys, EO sensing, as well as *in-situ* AUVs, ASVs, and static platforms that will offer the greatest potential to enhance our understanding of shelf sea ecosystem mechanisms (Box 1). Together, these approaches allow for drivers to be studied concurrently, in the locations and time scales most appropriate or least understood. However, the increasing demand for space in shelf seas due to the rapid expansion of OWFs, will introduce bias to several of these data sources. Spatial overlap between future OWF development and fishery surveys may result in changes to survey designs as well as reduce sampling effort, with subsequent implications on data quality (e.g. data gaps) that support our long-term understanding of such mechanisms (Haase et al. 2023, Methratta et al. 2023). On the other hand, OWF development presents an opportunity to encourage allocation of resources for assessment and monitoring efforts. Increasingly, the use of strategic networks of distributed ocean observatories interconnected by mobile platforms should be considered (Venkatesan et al. 2018). Such networks are possible at the regional scales of planned offshore wind (see the existing Lofoten-Vesterålen Ocean Observatory) as well as hypothetically within windfarms (Godø et al. 2014b). Placed at strategic locations and augmented with mobile surveys, the data generated (and shared) would have the potential to greatly enhance our understanding of key oceanographic processes and of relevance to industry, governments, and the wider scientific community (Camus et al. 2021). The growing stream of and need to efficiently integrate information from such sensors and platforms means that the concurrent development, adaptation, and application of automated techniques (e.g. machine learning, neural networks, artificial intelligence) for processing and filtering data will continue to be a priority (Beyan and Browman 2020).

Analysis: advances in ecological modelling

To be able to use detailed local data to predict at regional and shelf-wide scales, ecosystem modelling approaches that include the representation of drivers of ecosystem function at all scales are needed. While strides have been made in the last decade in the realm of Agent- and Individual-based Models (i.e. SeaBORD (Searle et al. 2018), iPCOD (Harwood et al. 2014), DEPONS (Nabe-Nielsen et al. 2018), these typically focus on one species of top predator at a time, are highly sensitive to dynamic changes in pelagic fish (prey) distribution (Searle et al. 2023) and may even be nonspatial (iPCOD). While fundamental niche and distribution models are ideal for the prediction of cumulative effects of wildlife responses and population dynamics in the face of multiple pressures (i.e. climate change, energy development, fisheries), these are constrained due to habitat heterogeneity and plasticity in animal abundance and behaviour across space and time (Matthiopoulos et al. 2022). However, as rapid yet robust predictions of the environmental effects of offshore wind are needed, ecological modelling offers valid alternatives. Mechanistic approaches include the parameter-rich food web modelling framework EwE: Ecopath with Ecosim that can be run in space (Ecospace) once different regions are identified (Nogues et al. 2023) or the 'end-to-end' functional group framework implemented in StrathE2E2 (Thorpe et al. 2022). Another even more promising approach is that of dynamic Bayesian Network (DBN) modelling, an extension of the well-established Bayesian Network technique for modelling time series (Friedman et al. 1999).

DBN is a parsimonious graphical modelling technique that can be used to capture ecological as well as spatio-temporal patterns between variables (Tucker and Duplisea 2012). Such probabilistic models allow predictions to be made across different spatial and temporal scales in response to stressors while simultaneously including a range of indicator species or functional groups to represent all trophic levels. Coupling physical dynamics from high resolution oceanographic models such as Finite Volume Community Ocean Model (FVCOM) into ecosystem models allows for critical habitat variables from local to regional and shelf-wide scales to be considered including physical (mixing and stratification: Chatzirodou et al. 2016, De Dominicis et al. 2018), biogeochemical (nutrients, oxygen: Tweddle et al. 2013, Hull et al. 2021), and ecological (plankton biomass and vertical distributions: Loveday et al. 2021) (Fig. 3). Crucially, dynamic Bayesian ecosystem models allows for predictions of both species-specific population trends at ecosystem-wide scales in different habitat types, as well as the main drivers of strong changes in any of these trends to be identified (Trifonova et al. 2021). Outputs from ecosystem models can be integrated into finer-scale models, such as niche and distribution models and the Habitat Risk Assessment (HRA) model (Declerck et al. 2022, InVEST: https://naturalcapitalproject.stanford.edu/). The use of such ecosystem model outputs as explanatory habitat variables has already enabled advances in the development of distribution models for higher trophic levels such as seabirds and marine mammals (Waggitt et al. 2018). For highly mobile, linked predator–prey species such as seabirds and fish, implementing Bayesian hierarchical joint models (using Integrated Nested Laplace Approximation: INLA) is now both robust and computationally efficient (Sadykova et al. 2017), allowing for high-resolution top-predator distributions to be predicted from the drivers (representing true cumulative effects) outputted from the DBN models (Box 1). DBNs can also readily be used to explore a range of 'what-if?' scenarios, based on potential changes in climate (e.g. temperature), OWF developments (e.g. stratification), and anthropogenic practices (e.g. commercial fisheries), as well as the specific trends (increases or declines) of different ecosystem components in response to these changes (Trifonova and Scott 2023). By providing an understanding of the reactive responses across and within all trophic levels, tractable predictions of the true dynamic nature of bottom-up (e.g. driven by temperature) versus top-down (e.g. driven by fishing) effects across trophic levels and habitats can be made (Trifonova et al. 2017). This, in turn, will allow for ecosystem-wide (true) cumulative effects to be predicted under multiple scenarios, at scales relevant to Environmental Impact Assessments and with assigned levels of confidence (Caro et al. 2020).

Implications and conclusions: priorities for studies

As the offshore wind industry continues to grow, there is a concurrent need for the scientific community to build the ecological evidence base so that practitioners can make informed decisions. Our North Sea case study demonstrates how to streamline this process through a hypothesis-driven investigative cycle that accounts for the inherent complexities of the marine environment (Box 1). Importantly, as shelf seas slated for increased OWF development differ in biophysical regimes and properties, it will be necessary to identify and measure indicators relevant to the specific region and/or basin of interest ('Problem/System' in Fig. 1).

Bottom-up forcing from 'physics to fish' (Sharples et al. 2013b) suggests that limited top-predator foraging locations are due to fish availability being tied to locations of new primary production as these are also limited areas where fish are actively foraging in space. As the introduction of OWF structures are suggested to be potential sites of new primary production and/or to promote large-scale changes of primary production, testing whether this is indeed the case at a local scale will be important in extrapolating effects at the shelf-wide scale (Dorrell et al. 2022). Furthermore, the type, configuration, and number of OWFs is likely to have varying impacts on spatially explicit levels of primary productivity due to the combination of local and regional changes to the available mixing energy (Daewel et al. 2022). As many fish species are linked to predictable seasonal changes in feeding and spawning grounds that are likely tied to locations of new primary productivity, investigating predictable variations in annual migration routes with monthly, seasonal and climate factors could provide predictions for top-predator distributions throughout the annual cycle and for the shelf-sea ecosystem, from which predictions on cumulative and combined effects of multiple wind farms can be made (Box 1).

The cumulative ecological effects of changes from OWF may impact how ecosystems function by pushing bio-physical variables and species interactions beyond natural variability; understanding how these changes interact with and impact/are impacted by the wider socio-economic landscape will also be critical (Methratta et al. 2020, Piet et al. 2021). Existing cumulative assessment frameworks (e.g. Cumulative Impact Assessment, CIA, Cumulative Effects Assessment) in theory allow for large-scale OWF effects to be evaluated in relation with other marine management sectors (e.g. fisheries) (Cavallo et al. 2017). However, these frameworks do not currently include dynamic links between different trophic levels, and interactions between stressors, thereby over-simplifying marine ecosystem processes and functioning (Willsteed et al. 2018). This contributes to uncertain assessments with a limited understanding of ecosystem-scale impacts to inform future OWF leasing rounds (Tweddle et al. 2018). It is therefore important to develop cumulative effects assessments to be able to integrate predicted ecosystem effects across the range of spatio-temporal scales at which changes can occur as shown can be done with the DBN modelling approach above and outlined in detail in Declerck et al. (2023).

The emergence of floating wind technology, that allows for extraction of wind energy further offshore in deeper more stratified waters, adds new challenges and opportunities for research (Farr et al. 2021, Lloret et al. 2022). Similar to fixed wind, floating developments will make alterations to localized wind (Wise and Bachynski 2020) with energy extraction creating wind wake effects, altering the degree of seasonal stratification, and but the differences in primary production may be opposite to those found at the shallower depths (< 50 m) of most static wind farms (Carpenter et al. 2016). As well, the difference in the depths of the main components of the substructures of floating wind turbines may affect plankton production differently due to the structures' movements increasing mixing within pycnoclines (Dorrell et al. 2022). Floating turbines have suspended cables in the water column, which may act differently to static turbine systems as Fish Aggregating Devices (FADs), attracting fish as well as providing many more surfaces within the whole water column for colonization by algae and invertebrates and other opportunistic species (Karlsson et al. 2022). However, new studies show differences between the sounds produced withing fixed and floating wind developments, with the moving components of the moorings creates impulsive and unpredictable sounds which may alter the expected FAD effects (Risch et al. 2023).

The expansion of floating offshore wind will also add to the spatial restrictions already imposed on fisheries due to fixed wind farms and other areas (Gill et al. 2020). There is a history of conflict between offshore wind and fisheries in space usage and socio-economic interests (Haggett et al. 2020); there is also concern about increased uncertainties in fish stock assessments due to displacement by offshore wind (Haase et al. 2023). Investigating the potential for co-location of floating turbines and a selection of fishing gear (e.g. creels, fish traps) has been identified as a research priority. The extent to which OWF developments act as de facto marine reserves (MPAs) that increase local fish stocks (Raoux et al. 2017), or complicate the achievement of conservation and biodiversity objectives (Lloret et al. 2023) has also yet to be determined; however, such effects can be predicted within the DBN modelling approach via what-if scenarios detailed above.

Moving forward, it will therefore be necessary to disentangle environmental effects from and investigate relationships between OWF development, concurrent climate change effects, and fisheries. Multitrophic, multiscalar, and above all hypothesis-driven studies rooted in ecological theory, recall the investigative cycle in Fig. 1, will be more important than ever, as these provide structure for the design of studies, counteract the increasingly untenable data mortgage scenario in marine science, and are adaptable to shifting baselines (i.e. climate change, emergence of new technologies). The resulting enhanced understanding of ecosystem-wide and cumulative OWF effects will be able to provide the empirical evidence-base to increase transferability of and certainty in CIAs, as well as inform marine spatial planning and management strategies.

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Supplementary data

Supplementary data is available at *ICES Journal of Marine Science* online.

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Data availability

The data underlying this article are available in the article and in its online supplementary material.

Author contributions

N.I., B.S., and B.W.: conceptualization; N.I., G.H., C.H., S.S., A.Z., M.D., K.G., and N.I.: investigation; N.I.: writing—original draft; N.I., B.S., G.H., M.D., K.G., C.H., S.S., N.T., J.J.W., J.U.W., C.W., A.Z., and B.W.: writing—review and editing; E.B., S.S., N.I., and B.S.: visualization; B.S. and B.W.: project administration; and B.S. and B.W.: funding acquisition.

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