Response
by the
Royal Society for the Protection of Birds

3 August 2012

Planning Act 2008

In the matter of:

Planning Application for construction of the Able Marine Energy Park on the South Bank of the River Humber at Immingham, North Lincolnshire

Planning Inspectorate Ref: TR030001
Registration Identification Ref: 10015550

1. In these representations, the RSPB continues to focus on the adequacy of the compensation at Cherry Cobs Sands. Other matters are addressed at the end of this document.

Compensation Sites

2. There are two inter-related issues: (1) the medium and long term quantum of inter-tidal mudflats which will be delivered at the Compensation Site; and (2) whether the quantum and quality of that compensation will properly compensate for the ecological function lost.

3. The 29th June Written Representations from the Applicant deal only with the first of these issues.

Quality and ecological function lost

4. Even if (which is denied) the "target" quantum of compensation is delivered in the medium term, it is plain that that quantum will not compensate for the ecological function lost for the reasons given in RSPB's Written Representations of 29th June. It is telling that there has been no attempt in the latest documents to explain how the compensation site will (over time) provide an appropriate *quality* of habitat to ensure the replication of the ecological function lost. The sole focus of the Applicant appears to be on the quantum. Even if the target quantum could be delivered, the Applicant still has to demonstrate and has not that the quality and quantum combined is such as to replicate the ecological function lost.

5. Further, the RSPB has serious concerns with the use of a 2:1 (or 1:1) ratio approach as if the adequacy of compensation was a purely mathematical question. The requirement is for the compensation to replicate the nature and extent of the ecological function lost. That turns on a detailed understanding of the existing function of the land to be lost and the functional links between it and other land and how that function and those functional links will be replaced. It is misconceived to proceed as if meeting a fixed ratio is all that is required. Annex I (attached) summarises previous ratio requirements for other managed realignment compensation schemes.

Quantum

6. Fundamentally, as the RSPB has been carefully explaining from the outset, the Applicant appear to accept (Ex 28.1 para 7.2.3 – para 7.2.5) that the current proposals will not deliver even what the Applicant to assess to be the necessary quantum of compensation and then even in just a five year window never mind the longer term. The Panel therefore now has Natural England, the RSPB, other statutory organisations and nature conservation organisations and the Applicant all agreeing that the current proposals will not deliver that which is required.

7. On the Applicant's own case it is therefore impossible to lawfully grant a Development Consent Order on the current proposals.

8. The RSPB is seriously aggrieved at the time and expense it has had to incur in responding to the current proposals when: (1) the RSPB has been clear from the outset as to the basic and fundamental flaws in the proposals; and (2) its concerns have (until this very late stage in the process) been ignored.

9. The existing proposals have been the basis of all the statutory processes to date. The Applicant say (EX 28.1 para 7.2.4) now that "alternative approaches" need to be, and are being, looked at. No details are provided and plainly the RSPB cannot comment on them. The statutory processes do not envisage or allow "alternative approaches" (particularly on matters so fundamental as the adequacy of compensation) being raised for the first time at this very late stage. The current programme necessarily will have to be stop and re-start when the Applicant has formulated alternative proposals for compensation which will replicate the ecological function lost in the medium and long term.

10. The RSPB has considered the new information provided in detail. There is nothing in the information which undermines the RSPB's earlier comments all of which remain valid. The following additional points are now made:

    a. The compensation required is principally inter-tidal mudflats not saltmarshes.

    b. The Applicant's analysis of the quantum of mudflats required is subject to significant (and in at least one respect unexplained) fluctuation: please see Annex II (attached).

c.   The modelling on the extent and rate of accretion is, of course, highly dependent on the height of the boundary between saltmarsh and inter-tidal mudflat. For the reasons given in Annex III (attached), that level is here +1.9m AODN level - above that level there is not just the possibility but the certainty that saltmarsh will develop and that inter-tidal mudflats will not exist.

d.   Applying the Applicant's own data sets as explained in Annex III, the inescapable conclusion is that the vast majority of the Compensation Site will develop into saltmarsh under the current proposals within 7 years. The RSPB's analysis of the Applicant's data demonstrates that at best (non-precautionary approach) there will be between 16.1 and 21.4ha of intertidal mudflat after just five years.

e.   It is therefore plain that even within the five year period there will not be delivery of anything like the quantum of compensation even the Applicant accept is required.

f.   In any event the Applicant modelling covers only a five year time frame. There is no evidential or scientific basis to assume that a stable state is reached in 5 years.

g.   No explanation appears to be provided as to the relevance of EX8.9 and EX11.24 (accretion at North Killingholme Marshes) for quantum and timing of compensation and the RSPB cannot therefore comment. Any possibility of long term (100 years) loss of parts of North Killingholme Marshes is legally irrelevant to the statutory question as to the adequacy of compensation now.

11.   Having reviewed all the further information, the RSPB considers that it confirms the analysis of Mark Dixon submitted with the RSPB's Written Representations (Annex C). The only addition required to his evidence in the light of the Applicant's representations is to add at the end of paragraph 6.1 (Annex C, the RSPB Written Representations):

"With mudflat forming below MHWNT of +1.9m ODN in this location, the predicted accretion will rapidly reduce the area of this habitat below the requirement accepted by Able."

12.   In addition to the points made above Appendix IV summaries the RSPB views on the Applicant's supplementary environmental Information (EX28.2) in relation to the proposed compensation site at Old Little Humber Farm.

The Way Forward

13.   Before any Development Consent Order can lawfully be made, the Applicant has to demonstrate on a precautionary basis that their compensation proposals compensate for the ecological function lost. That requires consideration of quantum and quality and timing[1].

14.   As the process to date demonstrates, the issue as to how that compensation can be delivered is not simply a matter of detail which can be put off for later consideration – it is fundamental to the grant of a lawful Development Consent Order and is not easy. It has to be grappled with and tested through the statutory processes.

15.   The package on offer does not replicate the ecological function lost for the reasons addressed in detail in RSPB's earlier evidence.

16.   Even if one just looks at quantum (a small part of the overall picture) it is now accepted that the proposals will not even meet the Applicant's understanding of the required quantum. As the Applicant appear to recognise it is therefore necessary to consider alternative proposals.

17.   The RSPB will engage with that process but not within the unrealistic timelines of the current programme. The statutory scheme plainly envisages that proposals are properly and fully worked up at the point they are submitted. The statutory processes do not envisage major changes to fundamental parts of a Nationally Significant Infrastructure Project proposal so late in the process. Further changes to the compensation proposals will obviously constitute "Further Information".

18.   The process must therefore stop, the Applicant to given time to formulate its alternative proposals and for the RSPB to be given time to comment before the process starts again at least in respect of the compensation.

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[1] On timing the compensation has to be in place and of the requisite quality at the time when the harm occurs and be provided long term to replicate the ecological resource which would otherwise have been available at NKM.

The Applicant's Comments on the RSPB's Relevant Representations

19. At page 162, paragraph 61.5 (of the Applicant's comments on the Relevant Representations documents submitted on 29ᵗʰ June 2012) the Applicant refers to the RSPB's involvement in the Immingham Agreement (attached at Appendix A to the Applicant's Comments on Relevant Representations document) and states in relation to the RSPB's Relevant Representations:

> *"The RSPB assert that, '(i)t is imperative that full details of both the mitigation and compensation measures are presented now in order for the application to be adequately considered by the IPC and enable it to comply with the Habitats and EIA Regulations'. Yet the same organisation was content to sign a document supporting the compensatory measures for Immingham Outer Harbour (refer to Appendix A, Schedule 2 of the Agreement) when the details of the compensatory proposals appear to have been unsupported by an ES."*

20. However the Compensation Agreement for Immingham Outer Harbour and Hull Quay 2005 was discussed and agreed before a planning application had been submitted for the compensation sites and therefore unlike the consideration of the Applicant's application here the consideration of compliance with the relevant environmental Impact assessment regulations was not relevant at that stage of ABP's proposal (please see paragraph 5.1 of the Compensation Agreement where it is acknowledged that further processes need to be complied with).

Annex I

Compensation ratios in some UK intertidal compensation cases

Context

1. To date, the approach to securing Article 6(4) compensation provision in the UK has been a combination of agreeing the impacts to be compensated (species, habitats, ecological functions) coupled with expert judgement on an appropriate package of measures to be secured by scheme proponents. The majority of Article 6(4) compensation schemes in the UK have involved direct or indirect impacts on intertidal habitats in relation to either port-related development or coastal flood defences.

2. In respect of compensation arising from port-related development, the majority of compensation schemes have been based on identification of key design parameters with statutory and voluntary nature conservation groups, and then working together to identify suitable areas of search, matching those parameters with land available on the market or, in the case of the London Gateway Harbour Empowerment Order, through targeted use of compulsory purchase.

Historic approach

3. Due to the paucity of habitat compensation schemes prior to the Habitats Regulations coming into force in 1994 (which meant no such schemes were available for evaluation), expert ecological judgement has been central to deciding:

    i.   The appropriate package of ecological measures to compensate for the predicted impacts.

    ii. The level of confidence in those measures succeeding.

    iii. Any adjustment to area of compensatory habitat required to address with uncertainty and risk (e.g. proximity, time-lag, ecological effectiveness). This manifests itself in the "compensation ratio".

4. As noted below, the main port-related compensation cases to date can be split in to two categories:

    i.   Direct loss of intertidal habitat.

    ii. Direct and indirect effects, in particular addressing temporary loss of ecological function.

Ratios

5. Based on this expert driven approach, early on the practice in the UK settled on a ratio of between 2:1 and 3:1 for direct, permanent loss of habitat from an SPA or SAC in cases where a definite adverse effect on site integrity was concluded. This can be seen in the cases of Immingham Outer Harbour and the original Quay 2005 scheme (now Green Port Hull) where the overall ratio was 2.2:1, and Bathside Bay Container Terminal with a ratio of 2.1:1. Each of these schemes was predicted to cause permanent loss of intertidal habitats.

6.	Lower ratios have been agreed in respect of cases dealing primarily with indirect and/or temporary effects where the impacted habitats were considered to retain some, albeit reduced, ecological function, but an adverse effect on site integrity could not be ruled out. For example, the ratio at London Gateway was 1.1:1, with the managed realignment schemes designed to deliver the compensation objectives in conjunction with the intertidal habitats affected.

7.	This overall approach has been broadly accepted by all developers to date, as well as the UK Government through its decisions confirming those compensation packages.

8.	Separately in respect of coastal flood risk management schemes the Environment Agency use the ratio of 3:1 to compensate for the direct and immediate loss of intertidal habitats to the footprint of flood defences.

Guidance

9.	There is relatively little guidance (policy or ecological) in the UK or EU on the appropriate approach to determining a correct ratio for compensatory measures. EC Guidance (EC, 2007, *Guidance document on Article 6(4) of the 'Habitats Directive' 92/43/EEC*) simply states that ratios should be well above 1:1. However, it then goes on to identify a series of criteria that need to be taken in to account in designing compensation measures. From these, and again using expert judgement, an appropriate compensation ratio could be derived.

10.	Based on its experience, the RSPB has developed a position on compensation provision that emphasises an approach based on replacing the ecological functions required to support the species and habitats affected. This deliberately eschews setting a hard and fast ratio in recognition of the need to assess the requirements on a case-by-case basis.

11.	It is only in recent years that empirical information on the effectiveness of intertidal compensation schemes has become available, due to time taken for these schemes to be implemented and monitored. For certain impacts, it is now apparent that reliance on a simplistic 2:1 or 3:1 ratio is inadequate, although it acts as a starting point for discussion. A more deliberative, functional approach is required.

Annex II

Inconsistencies between the Application Documents and the Supplementary Environmental Information

1. In EX11.24 there are numbers attached to various component of change but it is quite obscure as to where the specific figures arise. In none of the modelling in EX 8.8, 8.9 and 8.10 do the Applicant's consultants include any figures.

2. EX8.8 and 8.10 are of very limited value in assessing inter-tidal mudflat change as they were aimed at examining the sub-tidal and only undertaken for five periods up to a maximum of 30 weeks. Not the long term assessment claimed.

3. Figure AME-06033-G (EX11.24, Annex A) claims to identify all of the changes and provide the basis of the impact assessment, yet there appears to be no methodology for assessing either the delineation of zones on the map nor the numbers attached to them in the key to the Figure. In addition on this Figure there are areas of change identified and labelled 'postulated'. Either they have been modelled and the modelling should provide the basis for the locations and areas, or they should not be included as postulated can hardly be described as having scientific credibility. The postulation appears to be made by the Applicant.

4. Moreover, there are interpretations given which have not taken into account the functionality of some of the changes. For example, the figure given for mudflat creation at the breach of the re-alignment is 2ha in Tables 3-5 (EX11.24, pgs 16-18)(actually from their own data this is 1.8ha) but it is not recognised that mudflats in front of a breach will be subject to high water velocities and soft sediment will not be able to accrete there, so severely limiting the benthic invertebrates in that area. Hence it will not be fully functioning mudflat and it is methodologically incorrect to use it as offsetting the required compensation area. A full and justified explanation requires to be undertaken to overcome the opaqueness in the assessment.

5. EX11.24, paragraphs 31 to 36 (pgs 14-15) covers the use of a Defra and Environment Agency Report (2004)(the Report) and draws many erroneous conclusions to support the Applicant's case:

    i.   The use of the formula from the Report (p14)(is strictly for South/South West Britain and is not suitable for the Humber. Although that said, it is then used without identifying any errors that may be involved. Nor including an assessment of the many other parameters which may be relevant.

    ii. The incorrect use of the formula within EX11.24 (p14) then comes out with 2.54mAODN as being the limit for Spartina (which by co-incidence forms the argument for 2.5 being the limit in the Cherry Cobb compensation for saltmarsh development). Page 119 of the Report[1] clearly identifies the *Spartina anglica* lower limit is around MLWN.

    iii. Then EX11.24 argues that 450 tidal inundations is the limit for expected saltmarsh development (pg15) whereas the Report does not say that – it says 450-500 is the zone (not limit) of saltmarsh development, indeed under some circumstances saltmarsh can

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[1] Defra/EA (2004) Suitability Criteria for Habitat Creation – Report 1: Reviews of present practices and scientific literature relevant to site selection criteria. R&D Technical Report FD1917TR1. (see EX11.4, para 31) (see Annex II.A attached).

form and survive below that level) and below 500-600 inundations is the zone for expected mudflat development. However, the report clearly uses the norm for saltmarsh development to be MHWN.

6.   EX11.24, paragraph 36 then states that below 2.3mAODN saltmarsh is relatively constrained. While this is apparent from Paull Holme Strays (the fact saltmarsh formation had started at this level) it is shown that its development was very fast in the Humber so that after just 3 years it covered 34% of that original height zone.

Inconsistency of mudflat loss calculations

7.   EX 11.24 refers to an immediate loss of 29.5ha mudflat (Table 1, pg 9). It is presumed that this is calculated from the 31.5ha loss mentioned in the Environmental Statement (ES, chpt 11, pg 11-75, Table 11.17) and Habitats Regulations Assessment (HRA, chpt 5, pg 5-35, Table 5.6) minus the 2ha of mudflat created at the breach site (actually 1.8ha – see above).

8.   EX 11.23 refers to the functional loss of 11.6ha due to operational activities (based on a precautionary exclusion distance of 275m south of new quay (Table 1, Ref C and Appendix A, Figure AME 06077 rev B). In EX11.24, Table 1 this figure is added to the direct loss of 29.5ha to give a total of 41.1ha loss. At a ratio of 2:1 this results in a compensation figure of 82.2ha (EX11.24, pg 12 Table 2).

9.   This contrasts with the compensation figure of 76ha mentioned in the ES and HRA.

10. This functional loss of 11.6ha was not mentioned in the original submission documents. An 'Additional Functional' loss of 6ha is cited in Table 11.17 of the ES. The same figure is mentioned in Table 5.6 of the HRA, where it is described as 'Temporary functional loss' due to construction (Para 5.4.14).

11. The figures quoted in both Table 5.6 of the HRA and Table 11.17 of the ES are sourced from 'Chapter 2: Project Description' and 'Chapter 8: Compensation Measures'. It is not clear whether this is a reference to the ES or HRA, but in either case, no such chapters exist. This may be a reference to Chapters 4 (Description of the Development) and 9 (Compensation Measures) of the HRA, although these only mention the 'headline' figures of loss and compensation required (e.g. Paras 4.3.2, pg4-2, and 9.2.1, pg9-1).

Summary table

Loss of SPA mudflats (ha) Using the Applicant's terminology Author	ES Table 11.7 Chapter 11 Pg 11-75 ERM	HRA Table 5.6 Chapter 5 Pg 5-35 ERM	SEI - EX11.24 Table 1 Pg 9 AMEP
Direct	31.5	31.5	29.5
Indirect	-10.35 +7.88	+2	
'Temporary functional loss'		6	
'Additional functional loss'	6		
'Functional loss'			11.6
Total	39.97*	37.97	41.1

* minus 2ha created at compensation site (HRA, Para 9.2.1)

Suitability Criteria for Habitat Creation – Report I:

Reviews of present practices and scientific literature relevant to site selection criteria.

R&D Technical Report FD1917TR1

**Defra/Environment Agency
Flood and Coastal Defence R&D Programme**

Suitability Criteria for Habitat Creation – Report I:
Reviews of present practices and scientific literature relevant to site selection criteria.

R&D Technical Report FD1917TR1

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Table A: A summary of the criteria and thresholds relevant to habitat creation as derived from scientific reviews.

Criteria	Threshold	Habitat	Comments and References
Mean High Water Springs	MHWS – MHWN	Saltmarsh	Delineation by tidal level should be considered as a first approximation, and if possible, use site-specific information to give more accurate criteria.
Mean Low Water Springs			Thresholds for saltmarsh, e.g. Burd, 1989; S. Brown (pers. Measurements); Zedler, 1984; Webb and Newling, 1985; Woodhouse, 1979
Mean High Water Neaps	MHWN - MLWS	Intertidal flats	
Mean Low Water Neaps	Below MLWS	Eelgrass*	Thresholds for intertidal flats, e.g. Little, 2000; McLusky, 1989; Gray, 1981. Slope gradient thresholds from selected Environment Agency profiles of East Anglian intertidal flats (0.17 – 0.27%).
Elevation	Minimum at ~MHWN (450-500 inundations p.a.)	Saltmarsh	
	Lower than MHWN	Intertidal flats	General saltmarsh and intertidal flats texts, e.g. Adam, 1990; Packham and Willis, 1997; Long and Mason, 1983; Gray, 1992; Gray *et al*, 1995;
	Subtidal; MLWS to 4m	Eelgrass*	
Mean slope	1-2% (1:0-1:64) ideal. >0-7% (1:0-1:18) possible for saltmarsh	Saltmarsh Intertidal flats Eelgrass*	**NB**: Eelgrass is used to refer to subtidal *Zostera marina* only. Thresholds for eelgrass (*Z. marina*), e.g. Rodwell, 2000; Davison and Hughes, 1998
Length of site	Length of site along shore, parallel with waterline.	Saltmarsh Intertidal flats Eelgrass*	Parameters used to calculate overall area, from which habitat areas can be calculated based on inundation (see above for threshold information and references).
Width of site	Width of site across shore, perpendicular to waterline.	Saltmarsh Intertidal flats Eelgrass*	
Is the land polluted?	Absence of contaminants or presence below pollutant level	Saltmarsh Intertidal flats Eelgrass*	Assessment may be made using the EA's guidance: Contaminated Land Exposure Assessment (CLEA). Compare contaminant level measured at site with EA Soil Guideline Values and Groundwater and Contaminated Land publications: www.environment-agency.gov.uk/subjects/landquality/. Also public authorities hold Contaminated Land Registers and these should be consulted. Also for eelgrass see Davison and Hughes, 1998.
Water salinity	>10 – full salinity: optimum 22	Saltmarsh Intertidal flats	References, e.g.; Zedler, 1996.
	Saline	Eelgrass*	Almost exclusively in fully saline conditions in UK; e.g., Tutin, 1942; Stewart *et al*., 1994; Davison and Hughes, 1998

Water Quality	Absence of water-borne contaminants or presence below pollution levels e.g. EA Action Levels. Minor or no eutrophication/ nor elevated nutrients.	Saltmarsh Intertidal flats Eelgrass*	High levels of nutrients can produce algal blooms and mats, smothering invertebrate intertidal flats; e.g. Nicholls *et al*., 1981. Algae may also smother and kill saltmarsh vegetation; e.g. Adam, 1990; or eelgrass; e.g., Davison and Hughes, 1998; van Katwijk *et al*., 1997, 1999.
Light Climate	Tolerant of wide turbidity range.	Saltmarsh Intertidal flats	Turbidity levels affect composition of intertidal flats; e.g., Little, 2000.
	Intolerant of high turbidity, low light climate. Sensitive to physical disturbance.	Eelgrass*	Sensitive to turbidity and reduced light penetration; e.g., Giesen *et al*., 1990a & b; Duarte, 1991; Davison and Hughes, 1998.
Soil type	Various grain sizes from heavy clays to sands	Saltmarsh Intertidal flats	Grain size influences organic content and porosity affecting the competitive outcome of saltmarsh halophytes; e.g. Pye and French
	Sand – sandy/mud, sand/fine gravel	Eelgrass*	Reference; e.g., Davison and Hughes, 1998; de Jong *et al*., 2000)
Site Location	Muddy estuary with high accretion rates - resulting in potentially high rates of sedimentation. Open coastline with lower levels of suspended sediment is likely to accrete at a lower rate	Saltmarsh Intertidal flats	This parameter included to provide an indication of how likely it will be that the site evolves quickly due to settling of fine sediment. On the basis that it is unlikely that suspended sediment concentration levels will be known the options range from a muddy estuary (high suspended sediment concentrations) to an open coast (with lower SSCs)
Exposure	Sheltered, low energy environments protected from wave action	Saltmarsh Intertidal flats Eelgrass*	Penetration of high wave energy into the site will tend to inhibit settling of suspended sediment. Low currents and flows needed for eelgrass; Fonseca and Kenworthy, 1987; Fonseca *et al*., 1983; de Jonge *et al*., 2000
Freshwater flows	Freshwater can be a pollutant to habitats by reducing salinity	Saltmarsh Intertidal flats Eelgrass*	Saltmarsh/halophytic plants and eelgrass habitats polluted by reducing salinity. Diversity of marine organisms reduced in intertidal flats by freshwater. See 'salinity' references
Bed stability	Compressed soil is erosion-resistant. Weak, friable soil will erode more easily. Bed stability likely to increase with accretion, post breach	Saltmarsh Intertidal flats Eelgrass*	e.g. Whitehouse, Soulsby, Roberts and Mitchener (2000). Dynamics of Estuarine Muds. Thomas Telford Publishing.
Connectivity inside site	The degree to which a site drains will affect the proportion of intertidal flats to saltmarsh. Natural creek development in newly accreted material is slow - consider excavating channels pre-breach	Saltmarsh Intertidal flats Eelgrass	No references (T. Chesher, personal experience)

Propagule/ biological supply to site	Supply of seeds, rhizomes and tiller fragments needed to generate saltmarsh and eelgrass habitat, and supply of organisms for intertidal flats.	Saltmarsh Intertidal flats	Dependent on proximity of nearest established habitat and natural direction of transport; e.g., Koutsall *et al.*, 1987; Rand, 2000; Huiskes *et al.*, 1995; Garbutt *et al.*, in Reading *et al.*, 2002. Supply of larval or mobile adult invertebrates needed to generate intertidal flats communities; e.g. Little, 2000.
	Site needs to be directly adjacent to established eelgrass bed with identical environment.	Eelgrass*	Eelgrass growth in northern latitudes is thought to persist by vegetative means rather than seed production; e.g., Davison and Hughes, 1998; Fonseca *et al.*, 2000, 2002; Calumpong and Fonseca, 2001

***Eelgrass refers to subtidal *Zostera marina* only.

CONTENTS

A.Garbutt). A pioneer community in its European range, its decline in Britain is not fully understood but may be partly due to competition with *S.anglica*, and now seems to survive best where *S.anglica* is less aggressive, on drier sites above MHWS (Marchant and Goodman 1969a). It is at the northern limit of its range in Britain and small fluctuations in climate may also have played some part in its decline (Marchant 1967). Little viable seed is produced at the present time (Marchant and Goodman 1969a).

SM5 *Spartina alterniflora* salt marsh community

A naturalised alien, first recorded in Britain in 1829 from the river Itchen, Hampshire, and spread around Southampton Water east to Chichester (Marchant and Goodman 1969b). Hybridisation between the north American *S.alterniflora* and *S.maritima* produced *S.townsendii* and subsequently the fertile form, *S.anglica* (Gray *et al.* 1991). *S.alterniflora* now only survives at Marchwood, Hampshire, and as transplants in Poole Harbour, Dorset (Marchant and Goodman 1969b, Gray *et al.* 1990).

SM6 *Spartina anglica* salt marsh community

Constant species: *Spartina anglica*
Rare species: *Arthrocnemum perenne (Sarcocornia perennis)*

S.anglica (fertile form) arose from *S.townsendii*, a hybrid produced between the native *S.maritima* and the introduced North American *S.alterniflora*, which was first seen in Southampton Water in 1870. *S.anglica* was transplanted to many areas around Britain (and other parts of the world), and also spread rapidly around the coast. Large areas have died back since the 1930s, but it is widespread around the English and Welsh coasts, and still spreading around the Solway in Scotland.

S.anglica is found mainly at the seaward edges of salt marshes (lowest pioneer zone), and colonising old pans in the upper marsh zone. Substrates are varied, from very soft mud to shingle, although it appears to spread more on finer sediments. The pH is generally above 7.0 and loss-on-ignition (organics) varies from 0.2% to 36.3% (Adam 1976). The sediment is often strongly reduced (black layer), and *S.anglica* is very tolerant of tidal immersion, having colonised a 'vacant niche' in may areas, too low for other salt marsh plants to survive. In general its lower limit is around MHWN, with about 6 hours submersion per day during spring tides (Goodman et al. 1969, Dalby 1970, Morley 1973, Proctor 1980), but extends down to MLWN in the narrow tidal range of Poole Harbour, with up to 23.5 hours submersion per day on neap tides (Hubbard 1969). The lower limit of colonisation is probably controlled by wave or tidal action, and may also be related to the nature of the substrate. *S.anglica* can tolerate high salinities, up to about 2.5% chloride (Ranwell et al. 1964, Proctor 1980).

S.anglica spreads by rhizome fragmentation and seed, and small patches expand into clumps which may persist for long periods, or may spread and join together to form a continuous sward. Accretion of sediments in *Spartina* areas varies between 0.5 and 10cm per year (Ranwell 1964a, Bird and Ranwell 1964).

Annex III

Saltmarsh development

1. The definition of at what tidal level and how saltmarsh develops is a crucial part of the assessment of whether mudflat compensation proposed for the adverse functional impact of the Application on the SPA and Ramsar site, can be considered to be appropriate.

The level for saltmarsh development

2. I will look at the available evidence both within the Humber and wider in the UK, to see how ecologists have described and approached tidal levels associated with saltmarsh communities component plant species. I will concentrate on the lower levels of saltmarsh as that is the area of particular importance for assessing mudflat/saltmarsh interface. There are three measurements which are used by ecologists:

    i.   The relationship with the defined tidal levels (MHWS (Mean High Water Spring tides), MHWN (Mean High Water Neap tides) etc.) (the mean high water means the average of the water heights over the spring cycle periods).

    ii. The number of tidal inundations per annum.

    iii. The height Above Ordinance Datum Newlyn (AODN).

3. All three concepts are used within the Application and Supplementary Environmental information in different places.

The tidal level

4. Ecologically, this is the most meaningful, most widely used and incorporates the other two. It is a straight forward definition which holds true throughout a wide geographical range. While the other two appear more precise, they have many variables which need to be taken into account. That means that unless all relevant parameters are measured and incorporated, they are actually less clear as simple predictors of where mudflats may sustainably form.

5. Every estuarine ecologist works on the basis that MHWN is the boundary between mudflat and saltmarsh. This means that above this level everything either will be or is saltmarsh except where other factors expressly constrain it[1], though there is always a range of saltmarsh communities which develop above this from pioneer saltmarsh through mid saltmarsh to high saltmarsh. This last community may not be inundated on MHWS but would be on EHWS (Extreme High Water Springs) and on HAT (Highest Astronomic Tides).

6. In any assessment of the sustainability of mudflat the converse is required, ie levels which remain below MHWN, and are inundated on the vast majority of high tides.

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[1] Defra/Environment Agency (2004) Suitability Criteria for Habitat Creation – Report 1: Reviews of present practices and scientific literature relevant to site selection criteria. R&D Technical Report FD1917TR1. (see EX11.4 para 31). (see Annex I.A).

<u>The number of tidal inundations</u>

7.   The concept here is that the tidal level which experiences 450-500 tidal inundations a year will be the start of saltmarsh development or at around the level of MHWN. Conversely "Levels lower than this would be likely to develop mudflat" (Defra/EA 2004 see above). It should be noted that Ex 11.24 p 28 para 3, states that during neap tides the compensation site will not be flooded because of the invert level of the proposed breach is too high to allow tides to enter the site. This has profound implications for the improvement of conditions for the establishment of saltmarsh plants as well as the mortality of benthic invertebrates because most propagules of saltmarsh plants need a period of a few days without tidal cover when they can put out 'roots' to anchor themselves into the substratum whereas in extreme weather (hot or cold) exposed mudflats become much more hostile environments leading to increased invertebrate mortality.

<u>Elevation at mAODN</u>

8.   The use of mAODN is fraught with difficulties as for saltmarsh development it varies considerably around the UK and is a site specific measure and can vary depending on the grain size of the sediments that form the mudflat or saltmarsh, with, for example, sandy coarser sediment particles allowing saltmarsh plant establishment to lower levels through improved drainage characteristics. In Annex 32.5 of the Application, paras 3.1.3 – 3.1.6, there is a summary of the experience in relation to mAODN from the comparator site of Paull Holme Strays (PHS). It identifies the position as:

   i.   2.0 – 2.3mAODN *Spartina anglica* at the lowest levels within this band but with other species becoming apparent as the height (*in this range*) increases (Note that in the Paull Holme Strays monitoring, the 1.9mAODN was not separated and so the nearest level used was 2.0mAODN).

   ii.   2.3 – 2.6mAOD *Spartina anglica* is still dominant but a wide range of typical mid-level saltmarsh species becoming apparent.

   iii.   2.6 – 3.0mAOD the grass *Puccinellia maritima* is dominant with many other species.

9.   These saltmarsh communities are broadly consistent with the definition of low/pioneer, mid- and mid-high saltmarsh.

10.   In Ex11.24 para 7, the MHWN is given as +1.9mAODN. So, the conclusion that should be drawn is that above that level there is not just the possibility but the certainty, that saltmarsh will develop and mudflat will not exist.

Accretion and potential for saltmarsh development at Cherry Cobb re-alignment site

11.   Further information is provided on the accretion pattern and saltmarsh cover on (PHS) in Ex 11.24 Table 6, which now can be put into context of the start of saltmarsh development at the re-alignment site. This is compared with the information presented in Table 4 (Annex 32.5, page 7). I simplify these tables for clarity below:

Table 4: The average ground level (AODN) and accretion rates over the first five years at Paull Holme Strays. Table simplified from Table 4 in Annex 32.5.

Height mODN	September 2003	September 2006	September 2008
(in 2005)	Estimated level	+ 3 years	+ 5 years
2.0 – 2.3	1.82	2.33	2.46
2.3 – 2.6	2.18	2.51	2.60
2.6 – 3.0	2.77	2.85	2.89
3.0+	3.21	3.23	3.25

Table 6: The development of percentage of the intertidal areas in those bands cover of saltmarsh over the first five years at Paull Holme Strays. Table simplified from Table 6 in Ex 11.24.

Height mODN	September 2003	September 2006	September 2008
(in 2005)		+ 3 years	+ 5 years
2.0 – 2.3	0	0	1.5
2.3 – 2.6	0	1.3	34.2
2.6 – 3.0	0	44.1	76.8
3.0+	0	74.4	74.8

12.   It is important to note that all the heights considered, other than the estimated level in the lowest category in Table 4, are above MHWN and that all lower elevations had been raised substantially by accretion within 5 years. In terms of saltmarsh development, Table 6 shows that, even the lowest levels (from 2.0 – 2.3, ie MHWN and above) had started to develop saltmarsh within 5 years. However, it can be expected that the 1.5% cover of saltmarsh at the lowest level will very rapidly accrete in much the same way that the 1.3% cover in the 2.3-2.6mAODN band did between 2006 and 2008, so that within 7 years one would be expecting the 2.0-2.3 AODN band to be somewhere around 35-40% saltmarsh cover and just two years later, it would be around 75% cover of saltmarsh.

13.   The conclusion logically follows that the lesson from PHS is that the vast majority of the mudflats in the range MHWN to 2.6m initially created on the re-alignment at Cherry Cobb will rapidly develop into saltmarsh within 5-7 years. As summarised on pii in EA (2009) that *"Therefore, if the rate of accretion continues to be high, it is predicted that the majority of the site will become saltmarsh and that mudflat habitats will only persist in the areas close to the breaches."* [2] The modelling of the re-alignment has provided new estimates following the re-interpretation with new model parameters and this is replicated in Table A, from Section 5.1 in EX 28.1

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[2] Environment Agency (2009) Paull Holme Strays Environmental Monitoring Report. Part of the Humber Estuary Flood Defence Strategy (see Annex III.A attached).

Table A: Interpretation of Table 8 from EX 28.1.

Level mAODN	Scheme 1 ha	Area of height band	Pairs comparison	Scheme 2 ha	Area of height band	Pairs comparison
>3.0	105.4	5.1	Not comp	105.4	6.9	Not comp
2.5 – 3.0	100.3	59.2	66.5	98.5	55.2	64.9
2.4 – 2.5	41.1	7.3		43.3	9.7	
2.2 – 2.4	33.8	8.3	12.4	33.6	10.1	17.5
2.0 – 2.2	25.5	4.1		23.5	7.4	
1.8 – 2.0	21.4	7.4	21.4	16.1	5.8	16.1
<1.8	14.0	14.0		10.3	10.3	

14.   Here the data are used to show the amount of sustainable mudflat that would exist at all levels below 2.5mAODN, using that as the cut off. As shown above, no estuarine ecologist would ever accept that at a tidal elevation which is 0.6m higher than MHWN would be the cut off for sustainable mudflats. So, a correct ecological interpretation of the data presented for the two modelled schemes in Table A is that sustainable mudflats may develop only in the lowest two categories (ie 21.4 or 16.1ha) and it is likely that some of the mudflats of 1.9mAODN will develop saltmarsh in the longer term. The position after even just 10 years has not been presented, let alone the medium term defined by Able as 30 years.

The Humber Strategy
Managing the rising tides

Environment Agency

Paull Holme Strays
Environmental monitoring report

Part of the Humber Estuary Flood Defence Strategy

March 2009

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Figure 1.1:   General Plan of the Paull Holme Strays Site

Figure 2.1:   Aquatic Invertebrate and Water Vole Monitoring Locations

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SUMMARY

Background

Paull Holme Strays (PHS) is the site of the first major managed realignment scheme on the Humber. Created by the Environment Agency as part of the Humber Estuary Flood Risk Management Strategy, the site provides approximately 80 ha of new intertidal habitat and is fronted by the extensive Paull Holme Sands mudflat. It was initially anticipated that the PHS site would ultimately create approximately 45 ha of mudflat and 35 ha of saltmarsh. The site is adjacent to the Humber Estuary Special Protection Area (SPA), Ramsar site and candidate Special Area of Conservation (cSAC). These designations form part of the Natura 2000 network of 'European Sites' and illustrate the international importance of the estuary for, amongst other things, intertidal habitats and the wildfowl and waders they support. Nationally, the Humber Estuary is also designated as a Site of Special Scientific Interest (SSSI) for its mudflats, sandflats and saltmarsh habitats.

The main objectives of the PHS managed realignment project were to:

- Provide cost effective flood risk management for the area;

- Create intertidal habitat to compensate for that lost through implementation of this and other flood defence schemes in the middle estuary;

- Address additional habitat losses arising from coastal squeeze as identified in the Coastal Habitat Management Plan (CHaMP). These losses occur, when tidal defences prevent intertidal habitats migrating inland, in response to rising sea levels.

A five-year monitoring programme began in late 2003 to monitor the accretion and erosion at the PHS site and to assess the development of intertidal habitat and associated assemblages especially benthic invertebrates, birds and vegetation using complementary methods. The results of the fifth year of monitoring in the context of changes over the five year monitoring period are summarised in this report and will be used to inform management at the site as well as the design of further managed realignment projects in the Humber and elsewhere.

Accretion, Erosion and Vegetation

Accretion of sediments within the PHS realignment site was high and continued five years after breaching. Whilst the rate of annual increase had slowed since the first years after the breach, accretion at low elevations inside the realignment were still

three times higher than in less sheltered areas outside the site. At high elevations, rates inside and outside were comparable. Over the entire monitoring period from May 2004 to September 2008, the lower elevation areas of the site (survey stations at 2.0-2.3mODN in 2005) accumulated on average 47.1 cm of material. With back-calculated extrapolation to the time of the breach it was estimated that these areas have accreted 63.9cm of material. The development of drainage creeks has been observed to rapidly change the hydrodynamics of the site and suitability for vegetation development. Saltmarsh development has continued and vegetation has begun to colonise even the lowest areas of mudflat areas inside the site (Common Cord-grass *Spartina anglica*). By July 2008 vegetation covered 31% of the initially lower northern part of the site and 76% of the initially higher and part vegetated southern part of the site. Therefore, if the rate of accretion continues to be high, it is predicted that the majority of the site will become saltmarsh and that mudflat habitats will only persist in the areas close to the breaches. In July 2008, although relative abundance still differed, all saltmarsh species occurring outside the site also occurred inside the site and at equivalent elevations.

Intertidal Invertebrates

Benthic invertebrate communities continued to develop, although rates of change slowed, and in 2008 had achieved species richness comparable to the communities of the established mudflats outside the site, which remained relatively stable over time. However, there were still significant differences to the typical middle estuary community in terms of mean species richness, abundance and diversity. This is to be expected as benthic invertebrate communities have been observed to take longer than five years to develop elsewhere since rapidly accreting sediments are too fluidised for burrowing organisms to survive in. Inside the site the early colonising small bodied species present in high numbers (*Paranais litoralis*) were being replaced by less abundant larger bodied organisms (*Hediste diversicolor*). However, it is noted that the increase in saltmarsh and high elevations with low frequency of inundation are likely to reduce the supply of larvae required for colonisation. Terrestrial/freshwater organisms which dominated in 2004 are no longer a significant component of the community.

Birds

Development of the above benthic invertebrate community provides a valuable food source for waterbirds, and the bird assemblage present within the site in 2008 was considered broadly typical of a mid-estuary community. The mudflats of the northern part of the site supported the highest densities of foraging birds and the high water islets in the south of the site were important for roosting/loafing birds. Overall the inside of the site supported more foraging wildfowl and fewer waders than the areas outside the site and was functioning as an extension of the upper shore. Peak maxima of 174 Redshank *Tringa tetanus* and 365 Dunlin *Calidris alpina* were

recorded during the winter of 07 and 08. The site continues to be of international importance for Golden Plover *Pluvialis apricaria* in winter and Black-tailed Godwit *Limosa limosa* in spring and summer and also still supports a colony of Avocets *Recurvirostra avosetta*. However, after five years of intertidal habitat development, the overall density of the birds using the site was much lower than that at the sites lost; for which PHS is providing compensation.

Other surveys

The results of bird surveys undertaken by volunteers have broadly corresponded with the findings of the IECS bird surveys, and illustrate the changing usage of the site with early pioneer species such as Teal *Anas crecca* and Shelduck *Tadorna tadorna* numbers declining over time. Invertebrate fauna recorded also seem to have undergone some changes with time, however these species are particularly vulnerable to weather conditions and survey date. The replacement aquatic habitats built behind the new flood banks have been successful receptor sites for translocated water vole *Arvicola terrestris*, aquatic vegetation and aquatic invertebrates. Recovery from saline incursion during 2005 appears to be complete with few key taxa closely associated with brackish water still present in 2008.

Conclusions

The monitoring programme undertaken at PHS has proven to be a sound basis for assessing progress towards targets. Within five years the targets for habitat creation and usage by birds has been met, and benthic invertebrates are lagging slightly as expected at this stage. However, there is substantial evidence that the habitats and communities are continuing to change and that, if current rates of accretion continue, the end point for the realignment site is likely to represent a much higher proportion of saltmarsh to mudflat habitat than was initially planned. Strategic decisions are needed to plan for future management of the site and modifications to future plans at other sites in the Humber to ensure that the integrity of the European Sites is maintained in the long term. Recommendations include proposals for a further five years of monitoring at PHS, additional analyses to test hypotheses developed during the first five year reporting and the adoption of lessons learnt at this site into designs and monitoring programmes for managed realignment sites elsewhere.

elevation suitable for saltmarsh development. This site was quicker to develop saltmarsh than PHS, which may also have been contributed to by Freiston being surrounded by more extensive saltmarsh than PHS, with potential for a larger source of propagules. Propagule availability can be an important factor in limiting plant abundance within marsh zones (Rand, 2000; Wolters *et al.*, 2005). It has been estimated that the Freiston site may achieve an equivalent vegetation community to that outside after about 10 years post-breach (approximate estimate by S.L. Brown, in Brown *et al.*, 2007; Brown 2008).

4.5.8   Use of the Freiston realignment by fish showed that the new habitats were important nursery areas for juveniles, including commercial species (Brown *et al.*, 2007). Although not included in the monitoring programme, fish were sampled at PHS by IECS as part of the HARBASINS project. This showed that sand goby and flounder are the dominant fish along the Humber estuary, but that diagnostic species for middle/outer estuary sites including PHS were commercial marine species such as plaice, sole, whiting, sprat and seabass. Most of the fish caught in this study were small species or juveniles of species that rely on benthic food sources. This finding adds strength to the view of intertidal areas as important nursery grounds and essential habitats for estuarine and marine migrant species (Pérez-Dominguez, 2008).

4.6   Lessons Learnt

4.6.1   The design and monitoring of future managed realignment schemes in the Humber Estuary Flood Risk Management Strategy and elsewhere should be informed by the lessons learned and successes at PHS. As this is a relatively new management technique, guidance documents such as the CIRIA *Coastal and Estuarine Managed Realignment – Design Issues* handbook (Leggett *et al.*, 2004) have been devised based on limited real examples, therefore, as more empirical evidence becomes available such guidance can become more informed and the likelihood of achieving targets increased. A number of lessons from the PHS site are documented below and relate to design of the realignment for maximum conservation benefit as well as design and implementation of the monitoring programme.

4.6.2   A key point emerging from evaluation of the PHS monitoring scheme has been that physical parameters are key to the rates of development and sustainability of different habitat types, and that accretion rates can be in excess of 0.5m over five years where elevation is sufficiently low and sediment load and shelter are sufficiently high.

4.6.3   In particular, the results at PHS demonstrate the importance of elevation in fauna and flora community development and that saltmarsh development rate is proportional to time to achieve appropriate elevation. Due to changing

elevation with processes of accretion, it is recommended that, where available, comparison sites outside the scheme are distributed over a range of elevations to allow valid analyses to be undertaken. Further to this, it is also recommended that monitoring point locations are selected based on topographic data for the site prior to breaching. Such topographic data is likely to be useful in predicting which areas of the site may trap water and be slow to drain. Initial elevation data together with local accretion rates would help to refine predictive models used to assess the likely saltmarsh to mudflat ratio outcome of realignment sites.

4.6.4   Design lessons are also apparent when the results of the monitoring are reviewed in context with other sites, since sustainability of mudflat habitats is unlikely in an estuary with high suspended loads like the Humber without providing a greater degree of exposure. Very high deposition (several mm of sediment) can occur on the north Humber saltmarsh surfaces on a single high spring tide (Brown, 1998) in conditions of high turbidity, but much of it may be re-suspended on the ebb tide and carried away. Therefore, if mudflat habitat is desired in such locations, it may be desirable to remove a greater length of the former embankment with compensation for further loss of fronting saltmarsh, which may be inevitable with degradation of the seawall in the long term. If a strategic decision is made to stick to the original targets for mudflat creation at PHS in the shorter term, then remedial breach modification action may be required.

4.6.5   The erosion and topographic changes observed both at this site and at Frieston during the process of achieving hydrodynamic equilibrium (creek development and associated erosion) may be unavoidable, especially as there are cost implications in site preparation, breach excavation etc. However, the findings from these two sites suggest that more research into such change could help in determining optimum breach width, number of breaches, site gradient and configuration and density of starter creek systems. An early development of an efficient drainage system seems to be critical for the success of saltmarsh creation as creeks supply sediment and nutrients, dissipate tidal energy, and drain the marsh during the ebb tide. Good drainage increases sediment stability and reduces water-logging which is detrimental to plant and benthic fauna colonization and survival, and creek configuration is critical for enhancement of the important nursery role of saltmarsh for juvenile fish. Poorly drained sites can also be subject to lower accretion rates since more fluid sediment is more vulnerable to erosion (Brown *et al.*, 2007).

4.6.6   Monitoring of hydrodynamic changes may also help to inform possible designs to allow controlled warping directed at cost effective modification of unfavourable initial topography of a site. For example a phased approach to

warping and bank removal may rectify situations like the backward-sloping profiles at PHS, which may be an important long term sustainability issue when the fronting marsh is exposed to erosion after the remaining sea defence has eroded away.

4.6.7   Integration of monitoring and analysis for example of vegetation and accretion data has proved extremely successful in providing insights into species colonisation, establishment and succession of species on new areas exposed to tidal inundation. It is highly recommended that complementary monitoring is employed in all future monitoring programmes in order to increase the power of the data and to identify relationships and causal factors. However, the integration of analysis at PHS could be improved, since the annual reporting regime established at this site has not been amenable to full integrated analysis so that some hypothesised relationships e.g. between benthic fauna and accretion rate, have not been verified. This is largely related to contract programming and availability of funds, therefore, it is recommended that where possible arrangements for monitoring and analysis for the project lifecycle are in place prior to scheme implementation and that these are developed in light of circulation of all available data to all parties. This is particularly important to ensure that the rapid changes following breaching are captured, especially given that this scheme has demonstrated how important this factor is in explaining all other variables over the subsequent monitoring programme, and may also be used to refine outcome predictions.

4.7   Future Monitoring

4.7.1   The 2008 results show that all parameters measured within the intertidal habitats are still changing and/or have not yet become as diverse or of the same community structure as outside habitats. This provides a strong impetus for the continuation of monitoring at the site. At present we do not know how long it takes to create saltmarsh and mudflats with equivalent community structure to adjacent habitat, from good initial starting conditions. From a wider view, continued monitoring at a number of contrasting sites will enable estimates of progress against BAP targets to be calculated by proxy. In addition, there are few examples of monitoring of managed realignment sites over a long time scale (Halcrow, 2008). Therefore it is proposed that a further five year monitoring programme is undertaken, at the end of which, ten years of detailed managed realignment monitoring will have been recorded.

4.7.2   Due to the reduced rate of change, monitoring of accretion, vegetation and benthic invertebrate sampling is proposed to continue at the same level of detail, in order to retain scientific power, but at reduced frequency, i.e. twice

Annex IV

<u>**EX 28.2 Old Little Humber Farm**</u>

EX 28.2 has provided more details on the potential compensation area of Old Little Humber Farm (OLHF). The proposals fail to provide the RSPB with confidence that the area could be developed into a viable feeding area for Black-tailed Godwits as intended. The key points, in addition to those identified already in Annex B1 of the RSPB's Written Representations, which are lacking or, in the RSPB's extensive experience of creating wet grassland, will fail to deliver are:

1.   There is no detailed assessment of function which needs to be replicated by the compensation site.

2.   There is no detailed assessment of how the Black-tailed Godwits' ecology will enable them to take advantage of this site.

3.   It is appears that calculations on the water budget have been made for an average year; that is not appropriate when requiring to deliver a compensatory site based on the proposed short-term use of the site.

4.   Insufficient irrigation water will be provided due to evaporation from the water storage areas.

5.   The calculations on which the presumption is made that the site can provide enough macro-invertebrate food, are flawed.

6.   The proposals which are made to increase the macro-invertebrates on the site will not be successful in developing the biomass required.

7.   The timescale required for this site to deliver and the rapid colonisation of the Cherry Cobb re-alignment compensation area by saltmarsh, will mean that neither of the objectives to provide compensation for Black-tailed Godwits will be met.

<u>1. Assessment of required function</u>

The requirement of OLHF is to provide feeding grounds for up to 2,566 Back-tailed Godwits during the autumn months (July to October/November) while the re-alignment compensation site at Cherry Cobb matures.

Please note that the separation of the roost function of the Black-tailed Godwits at North Killingholme Haven Pits from the proposed feeding grounds at OLHF, has not been considered in developing this proposal.

<u>2. Assessment of the ecology of Black-tailed Godwit</u>

There are two main issues which emerge from the documentation. The first issue of the theoretical calculations has been covered in Annex B1 of the RSPB's Written Representations. However, the specific issues of soil penetrability appear not to have been adequately considered. The Godwits require soft soil in which to probe with their delicate beaks. It is recognised that, while levelling, the surface will be loosened (Para 3.2.14), but it is then suggested it should rolled in Para 3.5.5, which will decrease the penetrability of the surface. No assessment of required and observed penetrability

of the soil has been obtained or discussed. No information has been presented about the depths to which Black-tailed Godwits feed nor on which species of macro-invertebrates the Godwits are likely to feed. This should have been undertaken in order to design a site with the benefit of specific observations to fulfil the requirements of Black-tailed Godwits.

3. The water budget

It appears that the details of the water budget are provided in Report ref NABL 101/002/001 (see Para 3.1.1) but this report is not in the Applicant's Environmental Statement (ES) or the Supplementary Environmental Information. However, it can be surmised from Para 3.2.9 that the calculations have been made on the basis of an average year. Given the variation in rainfall over recent years as well as good practice, the calculations should have been made for a dry year to ensure that this habitat would provide for feeding Godwits. This would be the RSPB's approach to setting water budgets for grassland. This is even more critical when the intention is that this compensation site will only be in place for a 'temporary' period while the re-alignment matures; so it must work for all years.

4. Calculations on irrigation water

The proposals are for water in the storage areas to provide irrigation water for the Open Areas for the summer so that the latter are suitably wet for the autumn feeding function by Black-tailed Godwits. Para 3.2.17 identifies that the total run off which would be stored is 24,416m^3 and the irrigation requirement is 23,970m^3, a net surplus (in an average year) of 446m^3. The water storage areas total 4.77ha (EX28.2 Table 7).

We follow the information provided to us by the Applicant in NABL-101/002/002 (Thomson Ecology 2012 – see Annex IV.D attached) in Box 1 (pages 15-16). This uses the data from Climate and Drainage for Area 13, average annual rainfall of 655 mm, the driest quartile soil moisture deficit in dry grassland would be 105 mm by the end of August (page 43 Smith and Trafford 1976) (see Annex IV.C attached). From the calculations provided by the Applicant in section 5.2 of NABL-101/002/002 (see Annex IV.D attached) we cannot see where the evaporation from the water storage areas has been taken into account in assessing the actual water availability. It is a necessary rule of thumb, and one the RSPB would always use in wetland design, to ensure that there is sufficient water in most years; hence the driest quartile figure is used for calculations. So, the 105mm is the minimum equivalent loss that would be expected from open water bodies over the same time period as the irrigation need.

This would mean that 0.105m from the water storage areas of 47,700m^2 would be lost during the course of the summer. This amounts to a minimum of 5,008m^3 of water that will be lost from the water storage areas by evaporation alone. That means that instead of having a 446m^3 surplus, in a typical dry year there would be a minimum deficit of 4,562m^3 for irrigation – or a minimum shortfall of 19%. This is a minimum as evaporation from an open water surface is greater than the loss from dry grassland. So, the proposals as in EX28.2 would fail to provide enough irrigation water needed.

5. The calculations on the required macro-invertebrate biomass are flawed

This compensation site has a number of physical components but the only one which could potentially provide feeding conditions for Black-tailed Godwits, is the open grassland. There will be few, if any, earthworms in the water storage areas, even when they start to dry out (Figure 7 in Ausden *et al.*, 2001) (see Annex IV.A attached). Further, it is not clear if there will be shallow sloping sides to the storage areas or will they be steep sided. Table 3 (Appendix 1) calculates volume by multiplying water depth by area, which suggests that the sides will be vertical. The utility

embankments and the bunds for the water storage area will dry out completely during the autumn, so will be too dry for the birds to probe and thus will provide no feeding for the birds. This leaves us with just15.98ha of Open Areas which will provide feeding habitat for Godwits, which is less than half of the promised 38 hectares. (Appendix 1: Tables 7 & 8).

The objective to provide 40.35g/m^2 wet weight macro-invertebrate biomass was predicated on a 38ha site (paras 1.1.6 and 1.2.2). With the only grassland available being 15.98ha, it is clear that to hold the number of bird days calculated in Annex 35.6 (ES), the biomass required would be 38/15.98 x 40.35 = 96g/m^2 wet wt. Para 1.2.2 identifies that the current biomass is 7.59 g/m^2. Thus, to achieve this objective it will be necessary to increase the existing soil macro-invertebrate biomass to 12.6 times its current value. The required biomass of 96 g/m^2 is high - higher than the macro-invertebrate biomass found in eleven out of fourteen areas of permanent, unflooded lowland wet grassland nature reserves sampled by Ausden *et al.*, (2001) (see their Table 2) (see Annex IV.A attached). Importantly, even if it was possible to increase the biomass of soil macro-invertebrates to this high level, it would take several years to do so. For example, Barnard & Thompson (1985) (page 75) (see Annex IV.B attached) found that it takes about 5 years for soil macro-invertebrate biomass to increase to the levels found in permanent grassland following conversion of arable to grassland.

6. The proposals for increasing the macro-invertebrate biomass

The proposal is to put on up to 20 t/ha of organic matter to increase macro-invertebrate biomass (Para 3.2.14). This suggestion probably emanates from Treweek *et al* (1997: page 42) (see Annex IV.E attached) but there it relates to use in already existing grassland where there was already a high invertebrate biomass. Arable land holds a very low invertebrate biomass. EX28.2 provides no evidence of the timescale required to build up macro-invertebrate biomass to the required levels following conversion from arable to wet grassland. We have not been able to find any published evidence demonstrating that addition of organic matter accelerates the rate of increase in soil macro-invertebrate biomass during conversion of arable land to grassland.

7. The timescale for this site to develop in relation to the re-alignment compensation

The purpose of this site is to provide Black-tailed Godwit feeding areas while the Cherry Cobb re-alignment develops into functional mudflats capable of supporting the displaced Black-tailed Godwits. Given the long timescale required for any functioning wet grassland to achieve the required biomass of macro-invertebrates (see 6 above) and the fact that the re-alignment will be rapidly developing saltmarsh and so becoming unsuitable for feeding Black-tailed Godwits (see Annex III) within five years, the objective set for compensation from this site and the re-alignment, cannot scientifically be realised.

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The effects of flooding lowland wet grassland on soil macroinvertebrate prey of breeding wading birds

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Summary

1. Lowland wet grassland in western Europe is often managed for breeding wading birds, especially lapwing *Vanellus vanellus*, redshank *Tringa totanus*, snipe *Gallinago gallinago* and black-tailed godwit *Limosa limosa*. Recommended conservation management often entails introducing winter flooding, and in Britain there is government funding to encourage this through the Environmentally Sensitive Area scheme.

2. Soil macroinvertebrates are important prey for breeding wading birds on lowland wet grassland. This study quantified the response of soil macroinvertebrates to flooding, their ability to survive in flooded grassland, and changes in the abundance and physical availability of soil macroinvertebrates for feeding wading birds as flood water subsides.

3. Unflooded grasslands contained high biomasses of soil macroinvertebrates, comprising mainly Tipulidae larvae and earthworm species that are widespread in pastures. Grasslands with a long history of winter flooding contained much lower biomasses of soil macroinvertebrates, comprising mainly a limited range of semi-aquatic earthworm species.

4. Introducing winter flooding to previously unflooded grassland greatly reduced soil macroinvertebrate biomass. This was mainly due to the majority of earthworms vacating the soil soon after the onset of flooding. However, when earthworms were artificially confined in flooded soils, most species were capable of surviving periods of at least 120 days continual submergence. Winter flooding also expelled large numbers of over-wintering arthropods from the soil.

5. Soil macroinvertebrates were slow to recolonize winter-flooded grassland when it was re-immersed in spring. Consequently, prey biomass for breeding wading birds remained low in areas that had been flooded during the preceding winter. However, winter flooding probably benefited breeding snipe by helping keep the soil soft enough for them to probe for prey. It also probably benefited breeding lapwings and redshank by helping keep the sward short and open enough for them to feed in during the latter part of their breeding season. Pools of winter flood water that remained in spring and early summer also provided a source of aquatic invertebrate prey for breeding wading birds.

6. We suggest that the best feeding conditions for breeding snipe will be provided by keeping the upper soil soft enough for them to probe in but without reducing soil macroinvertebrate biomass by flooding it beforehand. Optimal conditions for breeding lapwings and redshank will probably be provided by creating a mosaic of unflooded grassland, winter-flooded grassland and shallow pools.

Key-words: ESA, flooding, habitat management, Lumbricidae, Tipulidae, wading birds.

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Introduction

Lowland wet grassland, defined as grassland below 200 m that is subject to periodic freshwater flooding or waterlogging, supports a distinctive assemblage of breeding wading birds in western Europe, including lapwing *Vanellus vanellus* (L.), redshank *Tringa totanus* (L.), snipe *Gallinago gallinago* (L.), black-tailed godwit *Limosa limosa* (L.), ruff *Philomachus pugnax* (L.), oystercatcher *Haematopus ostralegus* (L.) and curlew *Numenius arquata*

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(L.) (Beintema 1983; Smith 1983). Severe declines have been noted in the numbers of breeding wading birds on many areas of lowland wet grassland in recent years, and the plight of these species has attracted considerable concern from conservation organizations in Britain and the Netherlands. Overall, breeding lapwings have declined significantly, by 38%, between 1982 and 1989 on lowland wet grassland in England (O'Brien & Smith 1992), while the number of 10-km squares occupied by breeding snipe and redshank in the British Isles has declined by 22% and 12%, respectively, between 1968 and 1972 and 1988 and 1991, almost certainly due largely to the loss of suitable wet grassland (Gibbons, Reid & Chapman 1993). Black-tailed godwits and redshank also have an unfavourable conservation status in the rest of Europe (Tucker & Heath 1994).

Recommended conservation management for lowland wet grassland often entails the introduction of winter flooding. For the purpose of this paper, winter flooding is defined as the presence of surface water during periods between October and April inclusive. In Britain there is government funding to encourage patchy winter flooding through the Environmentally Sensitive Area (ESA) scheme. This is a system whereby landowners within designated areas receive incentive payments for entering voluntary management agreements. These agreements are intended to maintain or enhance the biodiversity value of each of the habitats for which the area is particularly noted, the quality of the landscape and its archaeological and historic resource (MAFF 1994). Similar schemes exist elsewhere in Europe. Of the eight English ESA that contain substantial areas of lowland wet grassland, six have management prescriptions intended to benefit breeding wading birds through the introduction of patchy flooding between the beginning of December or January and the end of April. These six ESA contain 46 000 ha of lowland wet grassland, representing about 23% of the English resource of this habitat (Dargie 1993; Glaves 1998).

Winter flooding attracts feeding and roosting wildfowl (Thomas 1982) and, by leaving shallow pools in its aftermath, provides suitable feeding habitat for breeding wildfowl (Thomas 1980) and wading birds (Green 1986; Ausden 1996a; Sanders 2000). Retention of winter flood water also enables field water levels to be kept high in spring and early summer. On peat soils, high field water levels are thought to increase the physical availability of soil macroinvertebrates for feeding snipe by reducing the penetration resistance of the upper soil, and thereby making it easier for them to probe (Green 1986, 1988; Green, Hirons & Cresswell 1990). Despite the introduction of winter flooding to many formerly unflooded grasslands, little is known of the effect that it has on the soil macroinvertebrates that are important prey for breeding wading birds on grassland, especially snipe (Hogstedt 1974; Matter 1982; Cramp & Simmons 1983; Green 1986, 1988; Galbraith 1989; Baines 1990; Green, Hirons & Cresswell 1990; Ausden 1996a).

It has generally been assumed that winter flooding does not adversely affect soil macroinvertebrates, because they are found in grasslands subject to regular and often prolonged periods of flooding (Green 1986, 1988; Green, Hirons & Cresswell 1990). Furthermore, the four most important sites for breeding snipe in England and Wales identified by Smith (1983), the Ouse Washes, Nene Washes, Lower Derwent Valley and the Somerset Levels, are all extensively flooded during winter.

Laboratory experiments have shown that the majority of the most abundant macroinvertebrates found in pastures, the earthworms *Allolobophora chlorotica* (Savigny 1826) (unpigmented morph), *Aporrectodea caliginosa* (Savigny 1826), *Aporrectodea longa* (Ude 1885) and *Lumbricus rubellus* Hoffmeister 1843, are capable of surviving for long periods (100–166 days) in aerated water (Roots 1956). However, the main physical stresses on earthworms in flooded soils are thought to be due to the development of anaerobic conditions in the soil (Mather & Christensen 1988). Hence the ability of earthworms to survive in aerated water cannot be used to predict their ability to survive in flooded soils.

In this study, we investigated the effects of flooding on the abundance and physical availability of soil macroinvertebrates to feeding wading birds. First, we compared the macroinvertebrate fauna of sites where water levels had recently been raised to that of traditionally winter-flooded grasslands that support important populations of breeding wading birds. Secondly, we carried out two experiments. The first investigated the relative preference of macroinvertebrates in flooded and unflooded soil. The second determined how long macroinvertebrates were able to survive in flooded soils. Finally, we investigated changes in the distribution of soil macroinvertebrates and their physical availability to feeding wading birds as flood waters subsided.

Methods

THE SOIL MACROINVERTEBRATE FAUNA OF LOWLAND WET GRASSLAND AND THE EFFECTS OF INTRODUCING WINTER FLOODING

The locations of sampling sites are shown in Fig. 1. We selected 12 sites where winter flooding had been introduced in the previous 1–14 years, and which had not been flooded prior to this for at least 20 years (see Table 2 for grid references). These represented most of the main sites where water levels have recently been raised by conservation bodies to benefit breeding waders in England, and are termed 'recently flooded grasslands'. All of the areas referred to in this paper were flooded with freshwater (salinity < 1‰ as determined using a hydrometer).

Ten of these recently flooded grasslands consisted of areas of uniform, agriculturally improved, grassland (National Vegetation Classification communities MG6 *Lolium perenne–Cynosurus cristatus* grassland or MG7

Fig. 1. Location of sampling sites.

Lolium perenne leys and related grasslands; Rodwell 1992). At each of these sites we selected a field that had been partially flooded the previous winter. At two sites, both agriculturally improved and agriculturally unimproved (National Vegetation Classification mire communities or MG5 *Cynosurus cristatus–Centaurea nigra* grassland; Rodwell 1991, 1992) fields were present. At these two sites we selected an agriculturally improved field and an agriculturally unimproved field, each of which had been partially flooded the previous winter. All the fields sampled had been partially flooded for periods of 20–120 days between October and March. Soil macroinvertebrates were sampled from unflooded and winter-flooded parts of the same field.

A further five sites were selected that had a long tradition (300+ years) of extensive winter flooding. These are termed 'traditionally flooded grasslands'. They were the four areas of lowland wet grassland identified by Smith (1983) as being the most important for breeding snipe in England: the Lower Derwent Valley, Nene Washes, Ouse Washes and Somerset Levels, and also the Insh Marshes (NH797020), the most important site for breeding snipe in Scotland (RSPB, unpublished figures). These sites also contained important breeding populations of other wading birds.

At the Somerset Levels, sampling was restricted to a hydrologically managed block on the RSPB's West Sedgemoor Reserve (ST360255), as this was the only area of the Somerset Levels that still held high densities of breeding snipe. At the Nene Washes, sampling was restricted to the RSPB Nene Washes Reserve (TF293997), as the rest of the Washes only supported low densities of breeding snipe. At the Ouse Washes, samples were taken from the Ouse Washes RSPB Reserve (TL490877), while at the Lower Derwent Valley samples were taken from Aughton Ings (SE697387) and Wheldrake Ings (SE443702), both parts of the Lower Derwent Valley National Nature Reserve.

Traditionally flooded grassland sites had estimated densities of breeding snipe of between 16·8 and 27·1 pairs km^{-2}. These compared with overall estimated densities of breeding snipe of 0·3, 0·9 and 3·3 pairs km^{-2} on lowland wet grassland found in three recent surveys of land within ESA (Robins, Smallshire & Street 1992; Weaver 1995; Allwood 1997).

At each traditionally flooded grassland, two flooded fields were selected that were considered typical of the site in terms of their vegetation, hydrology and use by breeding waders. The fields selected at these sites had been continually flooded for between 40 and 270 days during the winter (and autumn) prior to samples being taken.

Soil samples were dug and macroinvertebrates removed by hand-sorting and wet sieving (Gerard 1967; Ausden 1996b). Hand-sorting is considered the most reliable and effective method of sampling earthworms (Heppleston 1971; Edwards & Lofty 1972; Nordstrom & Rundgren 1972). Small and dark coloured worms, however, tend to be under-recorded using this method compared with chemical extraction (Raw

1959, 1960; Nordstrom & Rundgren 1972). However, as macroinvertebrate abundance was described in terms of biomass, rather than density, missing some small worms was unlikely to have a significant effect on the results.

Soil samples were dug with a spade using a quick, levering, action to intercept retreating large earthworms (Sims & Gerard 1985). Each soil sample was 20×20 cm in surface area and 10 cm deep. The depth of 10 cm was chosen as this is the approximate length of a snipe's and black-tailed godwit's beak, hence sampling would have removed only those soil macroinvertebrates within reach of feeding birds.

At recently flooded grasslands 12 randomly positioned soil samples were taken from unflooded parts of each field, and an additional eight randomly positioned samples from winter-flooded parts of the same field. At traditionally flooded grasslands, 12 randomly positioned soil samples were taken per field, except at the Lower Derwent Valley where, because of limited time, only eight samples were taken per field.

All but three sets of samples were taken between the first week of March and first week of April 1993–95 in order to measure macroinvertebrate biomass and availability just prior to waders arriving to breed. Samples from the Ouse Washes and West Sedgemoor had to be taken during the second half of May, as these sites were still extensively flooded until late spring. Samples from Insh Marshes were taken in 1992 using a similar methodology (Ausden 1992).

Soil samples were sealed individually in polythene bags, kept at 5 °C to prevent decay of any macroinvertebrates killed during digging, and sorted within a week. Soil macroinvertebrates were removed by first breaking the sample up by hand and removing any soil macroinvertebrates found. Any root mat that could not easily be broken apart by hand was then wet-sieved using a 2-mm gauge sieve and a high pressure water jet.

Soil macroinvertebrates were preserved in 4% formaldehyde solution and left for at least 3 days for their weight to equilibrate (Piearce 1984), blotted dry and weighed on a top pan balance to the nearest 0·01 g. Formaldehyde-preserved weight is about 25% less than fresh weight (Raw 1959). Invertebrates weighing less than 0·01 g were discarded as they were considered too small to be profitable prey for feeding waders and will in any case only have contributed a negligible proportion of the overall earthworm and tipulid biomass.

Soil macroinvertebrates were identified under a $\times 10$–40 binocular microscope. Earthworms were identified to species using the key in Sims & Gerard (1985). Nomenclature follows that of these authors. Although this key is for adult earthworms only, it also proved possible to identify the majority of immature and juvenile earthworms using features such as colour, shape, form of the prostomium, arrangement of setae and presence or absence of conspicuous dorsal pores. It was not possible to identify immature *Octolasion* spp. and some small immature *Lumbricus* spp. Also, some

fragments of earthworms, cut or damaged during taking samples, could not be identified to species. Unidentified earthworms only amounted to 3·5% of the total weight of earthworms found.

The most abundant Tipulidae larvae occurring on grasslands, *Tipula paludosa* Meigen 1830 and the widespread grassland species *Tipula oleracea* Linnaeus 1758, cannot be reliably distinguished using external features. Therefore, all Tipulidae larvae found (all of which resembled *T. paludosa* or *T. oleracea* as determined using the key by Brindle 1960) are referred to as *Tipula* sp.(p.).

Penetration resistance of the soil was determined using a penetrometer (Green 1988; Green, Hirons & Cresswell 1990). This measured the maximum force required to push a 4·5-mm diameter steel probe 10 cm into the ground. The diameter and depth of the probe were chosen to simulate a snipe's beak. Five measurements were taken adjacent to the location of each soil sample.

At six recently flooded sites, water table height was estimated by digging a temporary dipwell using a 2·5-cm diameter soil auger. The depth of the water table below the soil surface was measured at approximately half-hour intervals, until it had stopped rising.

RELATIVE PREFERENCES OF SOIL MACROINVERTEBRATES FOR FLOODED AND UNFLOODED SOIL

The relative preference of soil macroinvertebrates for flooded and unflooded soil was investigated by offering them the choice of flooded and unflooded halves of intact soil samples. Sixteen soil samples with a 20×40-cm surface area and a depth of 10 cm were dug from an unflooded agriculturally improved grassland on pelocalcareous gley soil at Buckenham RSPB Reserve (TG351053) in February 1994. Samples were sealed individually in clear polythene bags and stored in their original vertical orientation at a temperature of 10 °C for 6 days, to allow the soil samples to equilibrate to this temperature and any earthworms disturbed during transportation to re-establish their burrows. Eight of the soil samples then had their lower halves gradually immersed in rain water (also at 10 °C), which was trickled down the inside of the bags containing the soil samples, so that the water level within the bag was raised by 5 cm every hour. The bags were then resealed. The other eight control soil samples were opened and disturbed in the same way, but had no water added to them.

All bags were stored for 48 h at 10 °C and then each soil sample, while still contained within its polythene bag, was cut into upper and lower halves. A 50–100 g soil sample was taken from the centre of each half, sealed in a polythene bag and its available soil moisture content determined by drying to constant weight at 70 °C. The remainder of each soil sample was then also sealed separately in polythene bags prior to sorting. Macroinvertebrates were removed from the remainder

Table 1. Flooding treatments carried out on intact soil samples

	Treatment				
	0	60 (gradual)	90 (gradual)	120 (gradual)	120
Method of flooding	–	Gradual	Gradual	Gradual	Rapid
Timing of flooding	Left unflooded for 120 days	Flooded for 60 days and then left unflooded for 60 days	Flooded for 90 days and then left unflooded for 30 days	Flooded for 120 days	Flooded for 120 days

of each soil sample, preserved, identified and weighed as described in the previous section.

ABILITY OF MACROINVERTEBRATES TO SURVIVE IN FLOODED SOIL

The ability of macroinvertebrates to survive in flooded soil was investigated by artificially flooding soil macroinvertebrates in intact soil samples for different lengths of time. The timing and duration of flooding were chosen to mimic those prescribed under ESA management agreements.

Fifty intact soil samples each of 20×15-cm surface area and 20 cm deep were dug in December 1994 from the same area of Buckenham RSPB Reserve as described in the previous experiment. Samples were 20 cm deep to allow earthworms to avoid low winter temperatures in the surface soil during the course of the experiment (Gerard 1967). Each soil sample was sealed in a clear polythene bag to contain the invertebrates, and then reburied in the ground with its upper surface flush with the undisturbed soil surface. The soil samples were then left for a month to allow any earthworms disturbed during translocation to re-establish burrows.

During the first week of January the soil samples were flooded rapidly or gradually for up to 120 days using the treatments shown in Table 1. Each treatment was replicated 10 times. 'Gradual flooding' was intended to simulate flooding caused by deliberate raising of water levels. This was carried out by slowly pouring 200 cm³ of previously collected rain water down the inside of the bag containing the soil sample each day for 4 successive days until the soil sample became immersed. 'Rapid flooding' was intended to simulate flooding by infiltration of water from above, either as a result of heavy precipitation or through flooding caused by sudden large water inputs, for example due to a river overflowing. This was carried out by pouring rain water over the upper surface of each soil sample until, after a few seconds, it became totally immersed. Once each soil sample was immersed, further rain water was immediately added to cover it to a depth of 10 cm, so that the grass on the soil samples was completely submerged.

At the end of the flooding period, each soil sample was removed from the surrounding soil and immediately drained by making approximately 50 pin pricks in the bottom of the polythene bag. The unflooded soil samples also had 50 pin pricks made in the bottom of the polythene bag at the beginning of the experiment to allow them to gain or lose water at the same rate as the drained bags.

It was thought that containment of soil samples within polythene bags might cause the water above the soil to heat more than in a natural flooding situation. It was considered that this, together with the prevention of air and water movement within the sealed bags, might result in water in the upper soil becoming more rapidly deoxygenated than during natural flooding. To reduce these effects partially, all bags were opened at 2–5-day intervals for approximately 1 hour (not when raining) during the day to allow fresh air to enter. After the soil samples had been immersed for 4 days (the first day that the bags were checked) it was noticed that some arthropods had emerged from the soil and collected on the surface of the water and the insides of the polythene bags. These were collected and the bags were subsequently opened during the daytime at 2-day intervals to remove any invertebrates on the water's surface or on the bag. Any earthworm seen on the soil surface, in the water or on the insides of the polythene bags was also removed. This was because it was considered that they were not subject to flooding, and in a natural situation would either have migrated elsewhere or have been highly vulnerable to bird predation. Expulsion of arthropods virtually ceased after 10 days of flooding and, after this, bags were only opened approximately every 5 days to allow air to circulate and to collect any additional invertebrates from the water or insides of the polythene bags.

To determine whether conditions in the water in the polythene bags were similar to those under more natural conditions, an additional six soil samples were gradually flooded at the same time as the rest, buried and left with their polythene bags open. The temperature at the surface of the immersed soil samples was measured and compared with that of water above soil samples sealed in polythene bags, under different weather conditions on four occasions each month.

Maximum and minimum air temperature and minimum ground temperature were recorded daily at a weather station within 10 m of the buried soil samples throughout the course of the experiment.

Macroinvertebrates were removed from soil samples, preserved, identified and weighed as described previously.

CHANGES IN SOIL MACROINVERTEBRATE DISTRIBUTION AND PHYSICAL AVAILABILITY TO BREEDING WADING BIRDS AS FLOOD WATER SUBSIDES

Changes in soil macroinvertebrate distribution and physical conditions were measured along belt transects running perpendicularly across the margins of unflooded and winter-flooded grassland. Single transects were positioned in each of six partially flooded agriculturally improved fields at Buckenham RSPB Reserve (see previous experiments) and on pelo-alluvial gley soil at Church Farm Marshes RSPB Nature Reserve (TM465585). Each of the fields had been partially flooded for 90–120 days during the winter prior to sampling, and had been partially winter-flooded for the previous 1–5 years.

Transects were placed along a relatively straight section of flood margin. Soil macroinvertebrate biomass, vegetation height and physical conditions were measured along the transects in mid-March (immediately after flood water had begun to subside and just prior to waders settling to breed) and in mid-May, by which time lapwings and redshank have generally ceased feeding on soil macroinvertebrates and begun to feed mainly on terrestrial arthropods and/or aquatic invertebrates (Baines 1990; Ausden 1996a). Whittingham, Percival & Brown (2000) showed the importance of considering vegetation height for breeding waders.

Soil macroinvertebrate biomass, water table height, soil moisture, penetration resistance and vegetation height and cover were determined at 0·5-m intervals along the transects.

Soil macroinvertebrates were sampled by digging a 20 × 20-cm surface area and 10-cm deep soil sample dug at 0·5-m intervals along the transect. The vertical distribution of soil macroinvertebrates was investigated by dividing each sample horizontally into depths of 0–3 cm and 3–10 cm using a spade. The resulting two portions of the sample were separately sealed in polythene bags. Macroinvertebrates were removed from soil samples, preserved, identified and weighed as described previously.

Water table height, soil moisture and penetration resistance were determined using the methods described previously. Five measurements of penetration resistance were taken at each sampling point. Vegetation height and cover were measured using a point quadrat. This consisted of 10 1·2-mm diameter vertical graduated metal wires attached in a line at 5-cm intervals from each other. This was lowered vertically until it touched the soil surface. The maximum height at which vegetation touched each wire was recorded. Grass flower and seed heads were ignored, as these were unrepresentatively tall compared with the rest of the vegetation. If no vegetation touched the wire, then that point was recorded as bare ground. Two sets of 10 measurements were made at each sampling point.

DATA ANALYSIS

Assemblages of soil macroinvertebrates were ordinated in two axes using DECORANA (Hill 1979). All taxa were considered equally, i.e. there was no weighting for rare species. Data that were not normally distributed were transformed using the $\log(x + 1)$ or arcsine transformation to achieve normality prior to parametric tests being carried out (Sokal & Rolf 1969). Means are given ± 1 SE.

Results

THE SOIL MACROINVERTEBRATE FAUNA OF LOWLAND WET GRASSLAND AND THE EFFECTS OF INTRODUCING WINTER FLOODING

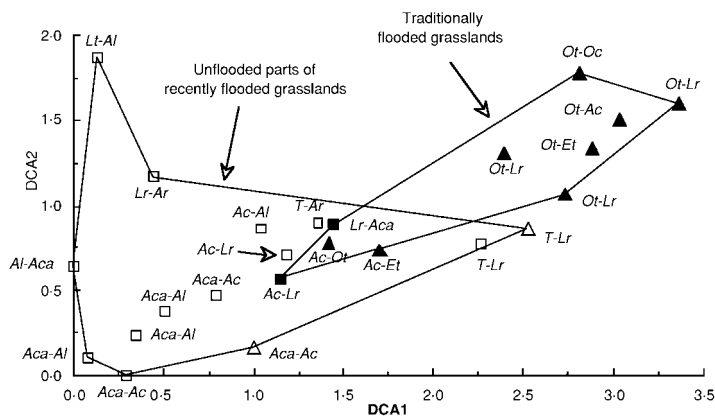Figure 2 shows the DECORANA ordination of soil macroinvertebrate assemblages from traditionally flooded

Fig. 2. DECORANA ordination of lowland wet grassland soil macroinvertebrate assemblages. Data are from traditionally flooded grasslands and unflooded parts of fields at recently flooded grasslands. Squares = gleys; triangles = peat. Black symbols = traditionally flooded grasslands; white symbols = unflooded parts of fields at recently flooded grasslands. The two abbreviations are the most abundant (left) and second most abundant (right) taxa in terms of biomass: *Ac*, *Allolobophora chlorotica*; *Aca*, *Aporrectodea caliginosa*; *Al*, *Aporrectodea longa*; *Ar*, *Aporrectodea rosea*; *Et*, *Eiseniella tetraedra*; *Lr*, *Lumbricus rubellus*; *Lt*, *Lumbricus terrestris*; *Oc*, *Octolasion cyaneum*; *Ot*, *Octolasion tyrtaeum*; *T*, *Tipula* sp.(p.).

grasslands and from unflooded parts of fields at recently flooded grasslands. Axes 1 and 2 of this ordination accounted for 70·8% of the total variance explained by the model. Soil macroinvertebrate assemblages from the unflooded parts of fields showed little, if any, overlap with those from traditionally flooded grasslands.

The majority of the soil macroinvertebrate biomass of traditionally flooded grasslands comprised the earthworms *Octolasion tyrtaeum* (Savigny 1826), *Allolobophora chlorotica* (green morph only), *Lumbricus rubellus* and *Eiseniella tetraedra* (Savigny 1826). These four species accounted for 95·5 ± 1·9% (range 90·4–100·0%) of the total earthworm biomass and 87·5 ± 3·2% (range 76·8–96·7%) of the total soil macroinvertebrate biomass at these five sites. The majority of soil macroinvertebrate biomass at three of the four peat sites comprised *Octolasion tyrtaeum*, but this species was not found in samples from the alluvial Lower Derwent Valley. The only specimens of *Aporrectodea caliginosa* were taken from an area of the Lower Derwent Valley a few metres from unflooded grassland. Their presence therefore may not be typical of traditionally flooded grassland. The only other species of earthworms found in traditionally flooded grasslands were *Dendrobaena octaedra*, *Octolasion cyaneum* (Savigny 1826) and *Satchellius mammalis* (Savigny 1826).

Some earthworm species were obviously able to withstand long durations of continual submergence. Four species, *Allolobophora chlorotica* (green morph), *Eiseniella tetraedra*, *Lumbricus rubellus* and *Octolasion tyrtaeum*, were found at the Ouse Washes, which had been completely submerged for approximately 270 days (between September and May) prior to sampling in late May. The Ouse Washes are frequently flooded for durations of more than 120 days during the winter and even for short periods during summer. *Octolasion cyaneum*

was present in the hydrologically managed block at West Sedgemoor, which had been flooded for approximately 150 days prior to sampling. *Dendrobaena octaedra* and *Satchellius mammalis* were only found in the Lower Derwent Valley and Insh Marshes, respectively, both of which had been continually flooded for periods of approximately 40 days during the winter prior to samples being taken.

In 11 of the unflooded parts of fields the majority of the soil macroinvertebrate biomass comprised the earthworms *Allolobophora chlorotica* (both green and unpigmented morphs), *Aporrectodea caliginosa*, *Aporrectodea longa*, *Aporrectodea rosea* (Savigny 1826), *Lumbricus rubellus* and *Lumbricus terrestris* Linnaeus 1758. These accounted for 93·9 ± 1·6% (range 84·0–100·0%) of the earthworm biomass and 87·6 ± 1·9% (range 74·5–96·4%) of the total soil macroinvertebrate biomass at these 11 sites. At Old Hall Marshes, Whiteslea and Elmley, the majority of the soil macroinvertebrate biomass comprised *Tipula* sp.(p.), being 72·4 ± 10·6%, 57·0 ± 14·2% and 57·6 ± 11·0% of the total soil macroinvertebrate biomass at these three sites, respectively.

At recently flooded grasslands, winter-flooded parts of fields nearly always contained significantly lower soil macroinvertebrate biomass than unflooded parts of the same field (Table 2). On average, winter-flooded parts of fields contained 9·8 ± 3·5% (range 0·0–44·5%, $n = 14$) of the soil macroinvertebrate biomass found in unflooded parts of the same field. Mean biomasses of six of the most abundant earthworm species, together with those of *Tipula* sp.(p.), Elateridae larvae and 'other Coleoptera larvae', were significantly lower in winter-flooded parts of fields than in unflooded parts of the same field (Table 3).

Total soil macroinvertebrate biomass in recently winter-flooded grasslands was significantly lower than in traditionally flooded grasslands (Table 3), and in the

Table 2. Total soil macroinvertebrate biomass in unflooded and winter-flooded parts of the same field. Data are for 14 fields at recently flooded grasslands. Years = approximate number of years that regular winter flooding has taken place. Soil: EP = earthy eu-fibrous peat; AG = pelo-alluvial gley; CG = pelo-calcareous gley. Veg: I = improved; U = unimproved. *t*-tests were carried out on log($x + 1$) transformed data. Biomasses are of formaldehyde-preserved specimens

Site	Years	Soil	Veg	Mean biomass ± SE (gm^{-2})		*t*	*P*
				Unflooded	Winter-flooded		
Berney Marshes TG472060	5	CG	I	82·20 ± 21·2	0·68 ± 0·65	5·407	< 0·0001
Buckenham TG348052	1	CG	I	47·15 ± 8·65	8·28 ± 5·20	4·356	0·0004
Castle Marshes TM474916	1	CG	I	109·08 ± 14·83	1·08 ± 0·88	10·767	< 0·0001
Church Farm Marshes TM464585	3	AG	I	37·73 ± 7·63	0·00	4·656	0·0002
Elmley TQ964674	14	AG	I	19·70 ± 4·35	0·30 ± 0·15	4·236	0·0005
Heigham Holmes TG440205	1	CG	I	178·20 ± 13·93	24·15 ± 6·70	9·901	< 0·0001
Hickling, field a TG425204	1	CG	I	56·35 ± 11·58	0·00	5·255	< 0·0001
Hickling, field b TG423206	1	CG	U	69·73 ± 20·63	3·45 ± 3·45	3·392	0·0032
Holkham TF878449	7	AG	I	158·45 ± 24·53	33·88 ± 17·73	3·841	0·0012
Old Hall Marshes TL974129	1	AG	U	28·65 ± 7·03	12·53 ± 7·95	1·763	0·0948
Strumpshaw TG342058	?	EP	F	44·73 ± 9·03	7·78 ± 2·58	3·739	0·0015
West Sedgemoor, field a ST377265	1	AG	I	41·55 ± 9·08	2·03 ± 1·10	5·096	< 0·0001
West Sedgemoor, field b ST348249	1	AG	U	41·25 ± 11·93	1·40 ± 0·53	3·925	0·0010
Whiteslea TG428217	3	EP	U	34·38 ± 12·70	1·63 ± 1·63	0·240	0·0240

Table 3. Biomass of soil macroinvertebrate taxa in lowland wet grasslands. Figures are mean formaldehyde-preserved biomass (g m^{-2}) $\pm$ SE. P_1 = probability of a significant difference between unflooded and winter-flooded parts of fields at recently flooded grasslands using paired *t*-tests; P_2 = probability of a significant difference between winter-flooded parts of fields at recently flooded grasslands and traditionally flooded grasslands using unpaired *t*-tests. *$P < 0.05$; **$P < 0.01$; ***$P < 0.001$. *t*-tests were carried out on log($x + 1$) transformed data

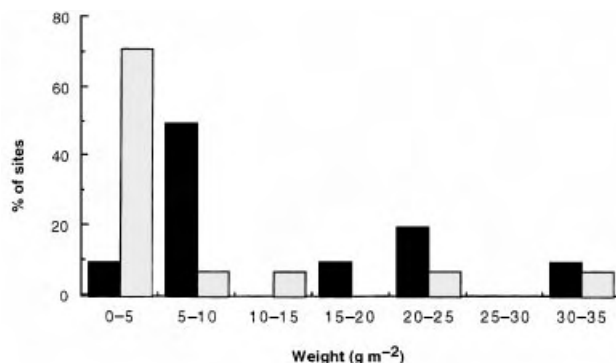| | Recently flooded grasslands | | | | Traditionally flooded |
Taxa	Unflooded ($n = 14$)	P_1	Winter-flooded ($n = 14$)	P_2	grasslands ($n = 5$)
All taxa	74·16 ± 13·46	***	6·50 ± 2·74	*	13·67 ± 2·96
Lumbricidae:					
Allolobophora chlorotica	10·66 ± 3·95	**	1·63 ± 0·61		4·23 ± 1·96
Aporrectodea caliginosa	17·11 ± 5·16	***	0·44 ± 0·44		0·00
Aporrectodea longa	15·14 ± 6·31	***	0·39 ± 0·27		0·00
Aporrectodea rosea	3·61 ± 1·12	***	0·00 ± 0·00		0·00
Dendrobaena octaedra	0·00		0·00		0·14 ± 0·14
Eiseniella tetraedra	0·70 ± 0·52		0·51 ± 0·34		1·00 ± 0·72
Lumbricus castaneus	1·28 ± 0·65	*	0·03 ± 0·03		0·00
Lumbricus festivus	0·40 ± 0·40		0·00		0·00
Lumbricus rubellus	6·49 ± 1·22	***	0·47 ± 0·29	*	2·29 ± 1·13
Lumbricus terrestris	1·76 ± 1·12		0·04 ± 0·04		0·00
Octolasion cyaneum	0·25 ± 0·25		0·00 ± 0·00	*	0·11 ± 0·09
Octolasion *tyrtaeum*	0·72 ± 0·42		0·00 ± 0·00	***	2·51 ± 0·75
Satchellius mammalis	0·02 ± 0·02		0·00 ± 0·00		0·12 ± 0·12
Immature Diptera:					
Tipula sp.(p.)	5·51 ± 1·93	***	1·03 ± 0·56		0·03 ± 0·03
Others	0·77 ± 0·20		0·48 ± 0·19		0·45 ± 0·22
Coleoptera larvae:					
Elateridae	0·52 ± 0·13	***	0·01 ± 0·01	*	0·16 ± 0·10
Others	0·69 ± 0·35	*	0·06 ± 0·04		0·17 ± 0·12

Fig. 3. Soil macroinvertebrate biomass in fields at traditionally flooded grasslands (black, $n = 10$) and in winter-flooded parts of fields at recently flooded grasslands (grey, $n = 14$).

majority of cases was less than 5 gm^{-2} (Fig. 3). Mean biomasses of *Octolasion tyrtaeum*, *Octolasion cyaneum*, *Lumbricus rubellus* and Elateridae larvae were all significantly lower in winter-flooded parts of fields of recently flooded grasslands than in traditionally flooded grasslands. It is noticeable that even the traditionally flooded grasslands that supported high densities of breeding snipe also had low soil macroinvertebrate biomass compared with unflooded grassland.

One of the intended benefits of raising water levels on lowland wet grassland is to decrease the penetration resistance of the upper soil so as to make it soft enough for snipe to probe for macroinvertebrates. Overall, there was no significant difference in median penetration resistance between unflooded and winter-flooded parts of fields at the recently flooded grasslands sampled

($T = 29·5$, $n = 13$, $P = 0·2632$). In four of the recently flooded grasslands, mean penetration resistance was significantly lower in winter-flooded parts of the field than in unflooded parts of the same field. In two fields it was significantly higher, and in eight fields there was no significant difference (Table 4).

Green (1988) found that penetration resistance of the soil tended to increase during the spring and summer, and that snipe stopped nesting once the mean penetration resistance of the soil in the vicinity of the nest exceeded *c.* 5·8 kg. In many recently flooded grasslands, both unflooded and winter-flooded parts of the field had a mean penetration resistance greatly in excess of 5·8 kg at the beginning of the breeding season, suggesting that they would have been too hard for breeding snipe to feed in.

Table 4. Penetration resistance in unflooded and winter-flooded parts of the same field at recently flooded grasslands. Soil: see Table 2. *Some sampling points were too hard for the penetrometer to probe

| Site	Soil	Mean penetration resistance ± SE (kg)			
		Unflooded	Winter-flooded	*t*	*P*
Berney Marshes	CG	10·2 ± 0·5	10·1 ± 0·6	0·160	0·8755
Buckenham	CG	6·5 ± 0·4	11·8 ± 0·5	9·142	< 0·0001
Castle Marshes	CG	9·8 ± 0·6	8·8 ± 0·4	1·330	0·2000
Church Farm Marshes	AG	8·9 ± 0·5	8·1 ± 0·4	1·272	0·2195
Elmley	AG	10·7 ± 0·4	9·9 ± 0·7	1·097	0·2870
Heigham Holmes	CG	4·9 ± 0·1	5·1 ± 0·3	0·666	0·5141
Hickling, field a	CG	10·2 ± 0·5	10·7 ± 0·8	0·578	0·5703
Hickling, field b	CG	9·8 ± 0·5	8·1 ± 0·4	2·683	0·0152
Holkham	AG	4·7 ± 0·7	4·2 ± 0·5	0·544	0·5930
Old Hall Marshes	AG	> 16·3*	9·3 ± 0·4	–	–
Strumpshaw	EP	3·8 ± 0·2	5·9 ± 0·4	5·745	< 0·0001
West Sedgemoor, field a	AG	11·0 ± 0·3	9·8 ± 0·3	2·265	0·0361
West Sedgemoor, field b	AG	11·3 ± 1·0	8·2 ± 0·3	2·740	0·0134
Whiteslea	EP	7·4 ± 0·2	6·5 ± 0·3	2·593	0·0184
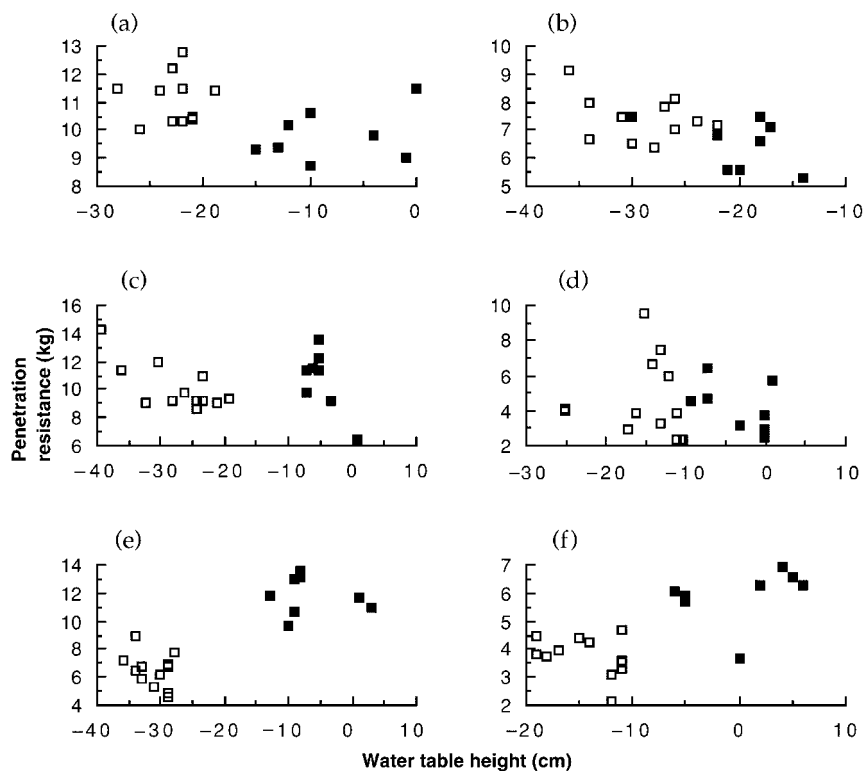
Fig. 4. Relationships between penetration resistance and water table height in different parts of the same field. Black squares = winter-flooded areas; white squares = unflooded areas. For water table height, positive values indicate the depth of water above the soil surface, and negative values the depth of water below the soil surface. (a) West Sedgemoor, site a (pelo-alluvial gley); (b) Whiteslea (earthy eu-fibrous peat); (c) Hickling, site a (pelo-calcareous gley); (d) Holkham (pelo-calcareous gley); (e) Buckenham (pelo-calcareous gley); (f) Strumpshaw (earthy eu-fibrous peat).

At the six sites where water table depth was measured, areas of fields that had been flooded during winter had a significantly higher median water table in early spring than unflooded parts of the rest of the field ($T = 0$, $n = 6$, $P = 0.0277$). Despite this, penetration resistance in winter-flooded areas could be lower than, similar to or higher than that in the rest of the field (Fig. 4 and Table 4). At Strumpshaw, visual inspections suggested that surface water only tended to remain on areas of the field that had a higher mineral content, and this was the most likely reason for the higher penetration resistance in the winter-flooded areas. At Buckenham the winter-flooded parts of the field had consolidated during flooding, and this probably increased its penetration resistance.

RELATIVE PREFERENCES OF SOIL MACROINVERTEBRATES FOR FLOODED AND UNFLOODED SOIL

Earthworms comprised 99·9% of the soil macroinvertebrate biomass in the 16 soil samples. For the four most abundant earthworm species, the percentage of their biomass in the flooded lower halves of the 'half-flooded' soil samples was significantly less than that in the lower halves of the unflooded controls (Table 5), indicating that these species strongly avoided flooded soil and would quickly vacate it following the onset of flooding. The flooded halves of the half-flooded soil samples had a significantly higher percentage soil moisture content than their upper halves (lower halves = 78·8 ± 5·2, upper halves = 54·8 ± 2·1, paired t = 4·423, n = 8, P < 0·0001). There was no significant difference in percentage soil moisture between the upper and lower halves of the unflooded controls (lower halves = 55·3 ± 1·8, upper halves = 52·9 ± 1·9, paired t = 1·015, n = 8, P = 0·344).

ABILITY OF MACROINVERTEBRATES TO SURVIVE IN FLOODED SOIL

An unexpected effect of the experimental flooding of soil samples was the emergence of over-wintering arthropods from them, particularly during the first 10 days of flooding (Fig. 5a). Significantly more arthropods

Table 5. Mean percentage of soil macroinvertebrates in flooded and unflooded lower halves of intact soil samples. For individual earthworm species, only those found in all 16 soil samples are shown. t and P are from t-tests carried out on arcsine transformed data

| Taxa	Percentage of the total soil macroinvertebrate biomass in each soil sample ± SE		t	P
	Unflooded lower halves of unflooded control soil samples	Flooded lower halves of half flooded soil samples		
All taxa	48·9 ± 3·6	8·4 ± 2·0	8·602	< 0·0001
Allolobophora chlorotica	41·0 ± 9·8	12·4 ± 4·0	2·934	0·0109
Aporrectodea caliginosa	49·7 ± 12·4	2·8 ± 2·8	4·335	0·0007
Aporrectodea longa	54·0 ± 3·1	9·1 ± 2·8	7·843	< 0·0001
Lumbricus castaneus	41·9 ± 7·3	1·3 ± 0·5	6·882	< 0·0001
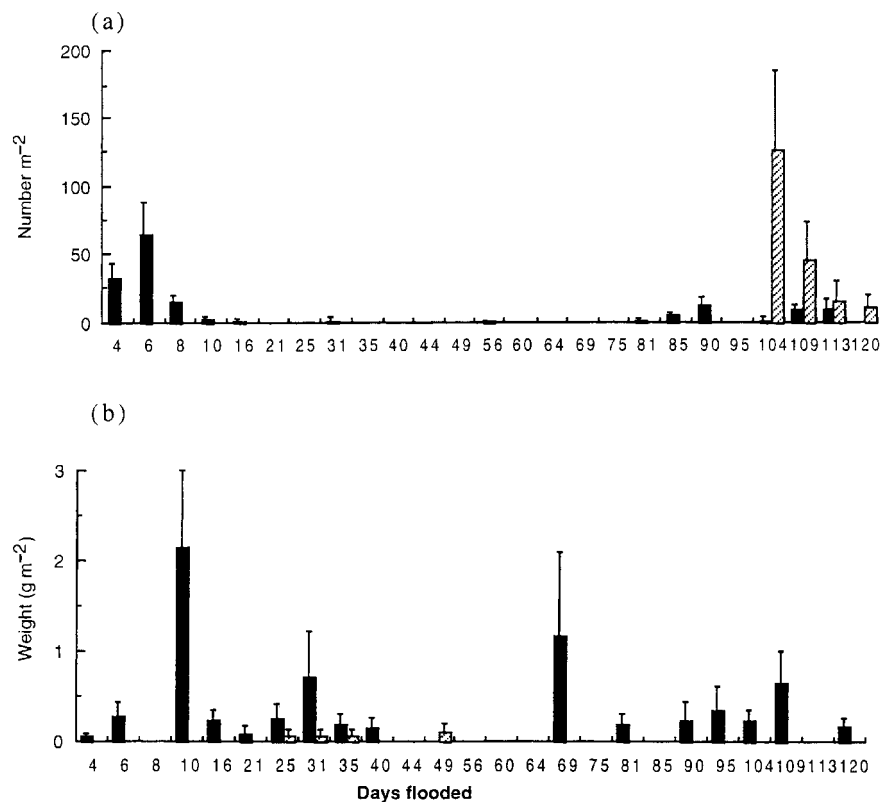
Fig. 5. Mean numbers of arthropods and biomasses of earthworms expelled from flooded (black) and unflooded (hatched) soil samples. (a) Arthropods; (b) earthworms. Sample sizes for flooded soil samples are 40 for days 0–60, 30 for days 64–90 and 20 for days 95–120. Sample size for unflooded soil samples is 10. Bars show ± 1 SE.

Table 6. Mean total numbers of arthropods removed from each soil sample during the experiment. Values are formaldehyde-preserved weight (g) ± SE. *F* and *P* are from ANOVAs performed on log(x + 1) transformed data. Means with different superscripts differed significantly from each other (Scheffe *F*-test, $P < 0.05$)

	Length of time flooded (days)						
Taxa	0	60 (gradual)	90 (gradual)	120 (gradual)	120	*F*	*P*
Arthropods other than adult Diptera	0·2 ± 0·1[a]	1·9 ± 1·0[ab]	4·5 ± 2·4[cb]	5·2 ± 2·3[cb]	2·4 ± 0·9[ab]	4·070	0·0080
Adult *Bibio* sp.(p.)	5·2 ± 2·3[d]	0·0[e]	0·0[e]	0·0[e]	0·0[e]	5·793	0·0010
Other adult Diptera	0·1 ± 0·1	1·7 ± 1·3	1·0 ± 0·3	0·3 ± 0·2	1·3 ± 0·9	1·321	0·2807

Table 7. Biomasses of soil macroinvertebrates removed from and remaining in flooded and unflooded soil samples. Values are formaldehyde-preserved mean weight (g) ± SE. *F* and *P* are from ANOVAs performed on log(x + 1) transformed data. Means with different superscripts differed significantly from each other (Scheffe *F*-test, $P < 0.05$)

	Length of time flooded (days)						
Taxa	0	60 (gradual)	90 (gradual)	120 (gradual)	120	*F*	*P*
Removed:							
All taxa	0·03 ± 0·01	0·15 ± 0·07	0·15 ± 0·10	0·30 ± 0·13	0·08 ± 0·05	1·486	0·2268
Lumbricus castaneus	0·03 ± 0·01	0·12 ± 0·06	0·11 ± 0·08	0·21 ± 0·10	0·06 ± 0·03	1·018	0·4112
Remaining:							
All taxa	7·05 ± 0·86[ab]	8·04 ± 1·14[a]	8·59 ± 0·94[c]	4·33 ± 0·87[bc]	3·39 ± 0·31[cd]	8·917	< 0·0001
Allolobophora chlorotica	0·72 ± 0·14	1·11 ± 0·16	1·13 ± 0·18	0·89 ± 0·18	0·85 ± 0·15	0·859	0·4980
Aporrectodea caliginosa	2·09 ± 0·40[ef]	2·60 ± 0·31[e]	1·60 ± 0·30[eg]	0·93 ± 0·26[fgh]	0·29 ± 0·15[h]	12·735	< 0·0001
Aporrectodea longa	2·66 ± 0·56	3·40 ± 0·82	4·01 ± 0·86	2·06 ± 0·52	1·99 ± 0·33	1·651	0·1828
Aporrectodea rosea	0·40 ± 0·13	0·23 ± 0·15	0·18 ± 0·08	0·23 ± 0·07	0·10 ± 0·06	1·727	0·1654
Lumbricus castaneus	0·72 ± 0·13[i]	0·69 ± 0·16[i]	1·05 ± 0·15[i]	0·12 ± 0·08[j]	0·05 ± 0·03[j]	19·831	< 0·0001

were collected from the water surface and insides of the polythene bags of the flooded soil samples during this 10-day period than from the soil surface and vegetation of the unflooded controls (flooded = 3·08 ± 0·87, unflooded = 0·00, $t = 3.377$, $P = 0.0015$). The majority of displaced arthropods were Staphylinidae (69·3%), Coleoptera larvae (12·1%), and Araneae (6·5%). Most adult Diptera collected from the flooded soil samples emerged during April, after between 85 and 115 days of flooding. Fifty-two of the 53 adult Diptera that emerged from the 10 unflooded soil samples during the second half of April and first half of May were *Bibio* sp.(p.). This represented a mean density of emerging adult *Bibio* sp.(p.) of 195 ± 87 m^{-2}. No adult *Bibio* sp.(p.) emerged from any of the soil samples that had been flooded (Table 6).

The only soil macroinvertebrates displaced from the soil by flooding were earthworms, and the majority of these were *Lumbricus castaneus* (Savigny 1826) (Fig. 5b and Table 7). Although lower biomasses were removed from the unflooded controls, there were no significant differences between the total biomass of soil macroinvertebrates removed from the different treatments.

Flooding resulted in little reduction in soil macroinvertebrate biomass. Only samples flooded for 120 days had a significantly lower biomass than the unflooded controls (Table 7). All of the most abundant earthworm species in the samples survived in flooded soil

for 120 days. Two species, *Lumbricus castaneus* and *Aporrectodea caliginosa*, showed possible reductions in biomass as a result of flooding, although the latter only occurred at a significantly lower biomass in one of the two treatments that were flooded for 120 days.

Comparisons of the temperature at the surface of the immersed soil samples suggested that the containment of soil samples within polythene bags did not greatly increase the temperature of the flood water compared with under 'natural' flooding conditions. The mean minimum and maximum ground temperatures during the experiment were 0·3 °C and 9·0 °C, respectively, for the first 30 days, increasing to 2·7 °C and 17·6 °C, respectively, for the last 30 days. Mean water temperature at the soil surface was only found to be significantly different ($P < 0.05$) between open and closed polythene bags on two occasions: early morning following a clear night during the first 30-day period (closed bags = 1·8 ± 0·1 °C, open bags = 1·3 ± 0·1 °C, $t = 3.411$, $n = 6$, $P = 0.0066$) and on a sunny afternoon during the last 30-day period (closed bags = 26·6 ± 0·5 °C, open bags = 22·7 ± 0·9 °C, $t = 3.680$, $n = 6$, $P = 0.0042$). This suggested that conditions in the polythene bags were similar to those under natural flooding. However, it was still possible that oxygen diffusion through the surface water from the air above was less than under natural flooding, because of restricted air and water movement within the sealed bags.

CHANGES IN SOIL MACROINVERTEBRATE DISTRIBUTION AND PHYSICAL AVAILABILITY TO BREEDING WADING BIRDS AS FLOOD WATER SUBSIDES

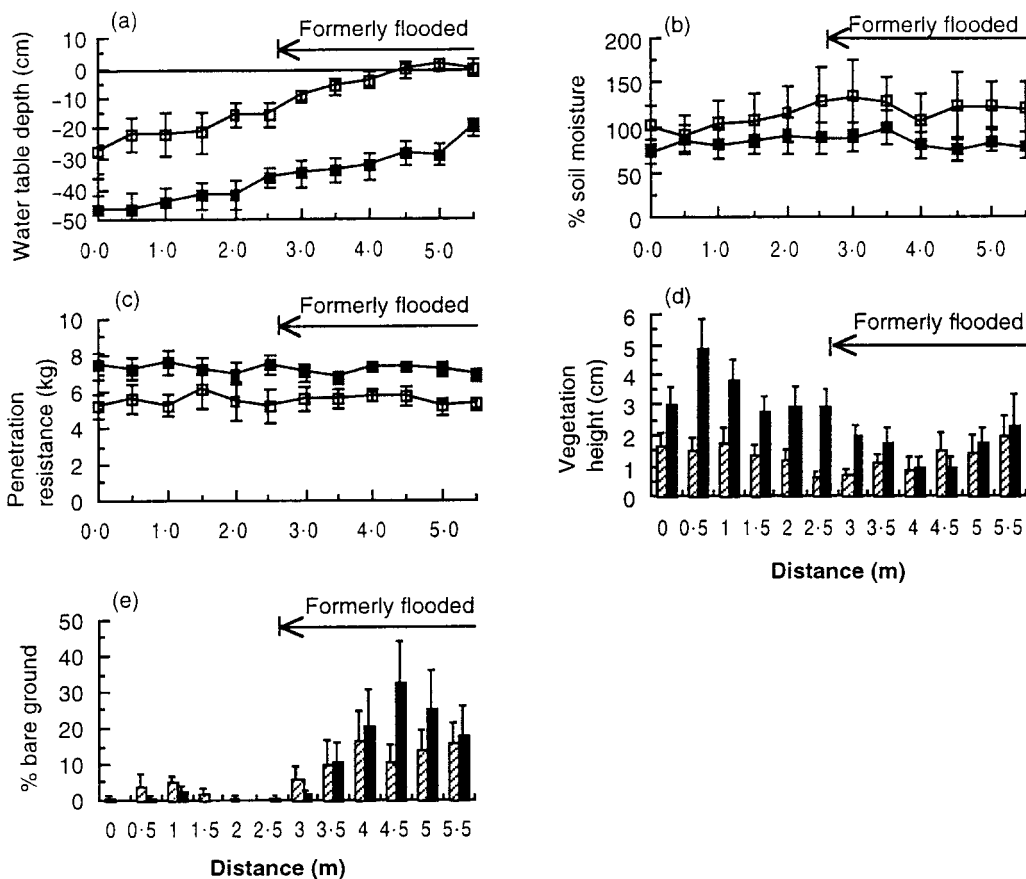Figure 6 shows physical conditions and vegetation height and cover along the six transects in March and May. Comparisons between mean physical conditions and vegetation height and cover along the entire lengths of the unflooded and winter-flooded sections of the transects in March and May are presented in Table 8.

As the flood water subsided, the water table fell significantly at each sampling point along the transects, falling relatively more towards the formerly flooded ends. Mean percentage soil moisture tended to be lower in May than in March, but these differences were not

Fig. 6. Mean environmental variables along six transects running perpendicularly across the margins of unflooded grassland in March (white squares or hatched bars) and May (black squares or black bars). (a) Water table depth relative to the soil surface; (b) soil moisture; (c) penetration resistance; (d) vegetation height; (e) percentage bare ground. Positive and negative values of water table height indicate the water table above and below the soil surface, respectively. Bars show ± 1 SE.

Table 8. Environmental variables and vegetation height and cover in unflooded and winter-flooded grassland. Data are from six transects running perpendicularly across the margins of unflooded and winter-flooded grassland. Unfl. = unflooded half of the transect; W. fl. = winter-flooded half of the transect. Paired *t*-tests on water table depth, percentage soil moisture and vegetation cover were performed on $\log(x + 1)$ transformed data

Variable		Mean ± SE		*t*	*P*
		March	May		
Water table depth below the soil surface (cm)	Unfl.	−21·4 ± 6·0	−43·0 ± 4·4	−2·444	0·0013
	W. fl.	−3·3 ± 2·1	−29·9 ± 3·1	−6·072	0·0018
Soil moisture (% of dry weight)	Unfl.	109·0 ± 26·8	84·5 ± 16·3	2·351	0·0655
	W. fl.	123·9 ± 30·5	84·6 ± 14·2	2·290	0·0706
Penetration resistance (kg)	Unfl.	5·48 ± 0·84	7·31 ± 0·62	−3·755	0·0132
	W. fl.	5·51 ± 0·36	7·09 ± 0·23	−2·982	0·0307
Vegetation height (cm)	Unfl.	1·35 ± 0·31	3·42 ± 0·54	−8·278	0·0004
	W. fl.	1·27 ± 0·36	1·63 ± 0·27	−0·726	0·5004
Bare ground (index out of 100)	Unfl.	1·95 ± 0·80	0·83 ± 0·43	0·738	0·1497
	W. fl.	12·22 ± 4·16	19·72 ± 5·36	−0·889	0·4147

quite significant along each half of the transects. Mean penetration resistance was significantly higher along the unflooded and formerly flooded lengths of the transects during May than it had been during March.

During March the vegetation was consistently short along the whole lengths of the transects. The fields had been heavily grazed by cattle and sheep the previous year to produce suitable conditions for grazing wildfowl, particularly wigeon *Anas penelope* (L.), which in turn had maintained a short sward over winter. The sward was also relatively open along the transects, there being a high proportion of unvegetated ground, particularly in the areas that had been flooded. By May the vegetation had grown significantly higher along the unflooded lengths of the transects and on the extreme margins of the winter-flooded grassland. These areas were dominated by vigorous, agriculturally productive, grass species, particularly *Lolium perenne* (L.). The formerly flooded lengths of the transects largely comprised the low-growing grass *Agrostis stolonifera* (L.). There were no significant changes in the percentage of bare ground between March and May.

Changes in soil macroinvertebrate biomass along the transects between March and May are shown in Fig. 7. Mean total soil macroinvertebrate biomass remained low along the formerly flooded sections of the transects in March and May, although it was slightly higher within a metre of the former flood margin. This distribution was shown by most taxa: *Allolobophora chlorotica*, *Aporrectodea caliginosa*, *Aporrectodea longa*, *Aporrectodea rosea*, *Lumbricus castaneus*, *Tipula* sp.(p.) larvae and 'other immature Diptera'. The highest biomass of the semi-aquatic *Eiseniella tetraedra* was around the former flood margin, but this species still only occurred at low biomass within the winter-flooded grassland further than 0·5 m from the former flood margin. *Tipula pierrei* (Tonnoir in Goetghebuer & Tonnoir 1921) larvae, Eristaline larvae and Chironomidae larvae were only found in the formerly flooded grassland.

There were no significant differences in total soil macroinvertebrate biomass between March and May at any sampling point along the transects, or along the unflooded and formerly flooded lengths of the transects as a whole (Table 9). The only taxa to show significant differences in biomass were *Eiseniella tetraedra* and the main aquatic immature Diptera. The only remaining individuals of the latter found in May were buried Eristaline pupae.

The majority of soil macroinvertebrate biomass in the top 10 cm of soil was within 3 cm of the soil surface, both during March and May (Table 10). *Eiseniella tetraedra*, *Lumbricus castaneus* and all immature Diptera including *Tipula* sp.(p.) were more or less restricted to within 3 cm of the soil surface. *Aporrectodea longa* and *Aporrectodea rosea* were mainly 3 cm or more below the soil surface. *Allolobophora chlorotica* and *Aporrectodea caliginosa* showed vertical distributions intermediate between these two extremes. Two species, *Aporrectodea longa* and *Eiseniella tetraedra*, appeared

to retreat from the soil surface as the season progressed. *Aporrectodea caliginosa* showed a similar, but not significant, trend.

Discussion

THE SOIL MACROINVERTEBRATE FAUNA OF LOWLAND WET GRASSLAND AND THE EFFECTS OF INTRODUCING WINTER FLOODING

Grasslands with a long history of winter flooding have a markedly different fauna to that of unflooded grasslands where attempts have been made to raise water levels to benefit breeding wading birds and other waterfowl. Although lowland wet grassland soil macroinvertebrates are capable of surviving long periods of flooding, this ability is restricted to just a few species. The most important of these in terms of biomass are the earthworms *Allolobophora chlorotica*, *Eiseniella tetraedra*, *Lumbricus rubellus* and *Octolasion tyrtaeum*. Various combinations of these four species have also been found to comprise the majority of the soil macroinvertebrate biomass in other winter-flooded peat and mineral soils (Cotton & Curry 1980; Baker 1983; A.L. Reid & T.G. Piearce, unpublished data).

Of the four most abundant earthworm species capable of surviving long periods of flooding, two species, *Allolobophora chlorotica* and *Lumbricus rubellus*, were also both common in unflooded grasslands. *Lumbricus rubellus* is found in a wide range of habitats, but mainly those with a high organic and soil moisture content (Sims & Gerard 1985). The green morph of *A. chlorotica* is typical of wet soil and is replaced by its unpigmented form in drier conditions (Satchell 1967a; Sims & Gerard 1985). The other most abundant earthworm species in traditionally flooded grassland, *Octolasion tyrtaeum* and *Eiseniella tetraedra*, were less frequently found in samples taken from unflooded grasslands. *Eiseniella tetraedra* and *Octolasion tyrtaeum* are capable of surviving for long periods underwater, and are regularly found in rivers (Sims & Gerard 1985). Both species possess quadrangular caudal ends that they move in the water to maintain gas exchange (Bouché 1970). *Octolasion tyrtaeum* is thought to survive flooding better than other widely distributed soil-inhabiting earthworm species, due to its well-developed subcutaneous net of blood vessels and high concentrations of haemoglobin, which enable it to inhabit badly aerated soils (Perel 1977).

The earthworm-dominated fauna of the unflooded grasslands was similar to that of other lowland pasture (Guild 1951; Gerard 1967) but differed in the presence of the green, rather than unpigmented, morph of *Allolobophora chlorotica* and in the lower abundance of *Lumbricus terrestris*.

The biomass of most soil macroinvertebrate taxa was greatly reduced in areas subject to winter flooding, although it was clear that several species of earthworms
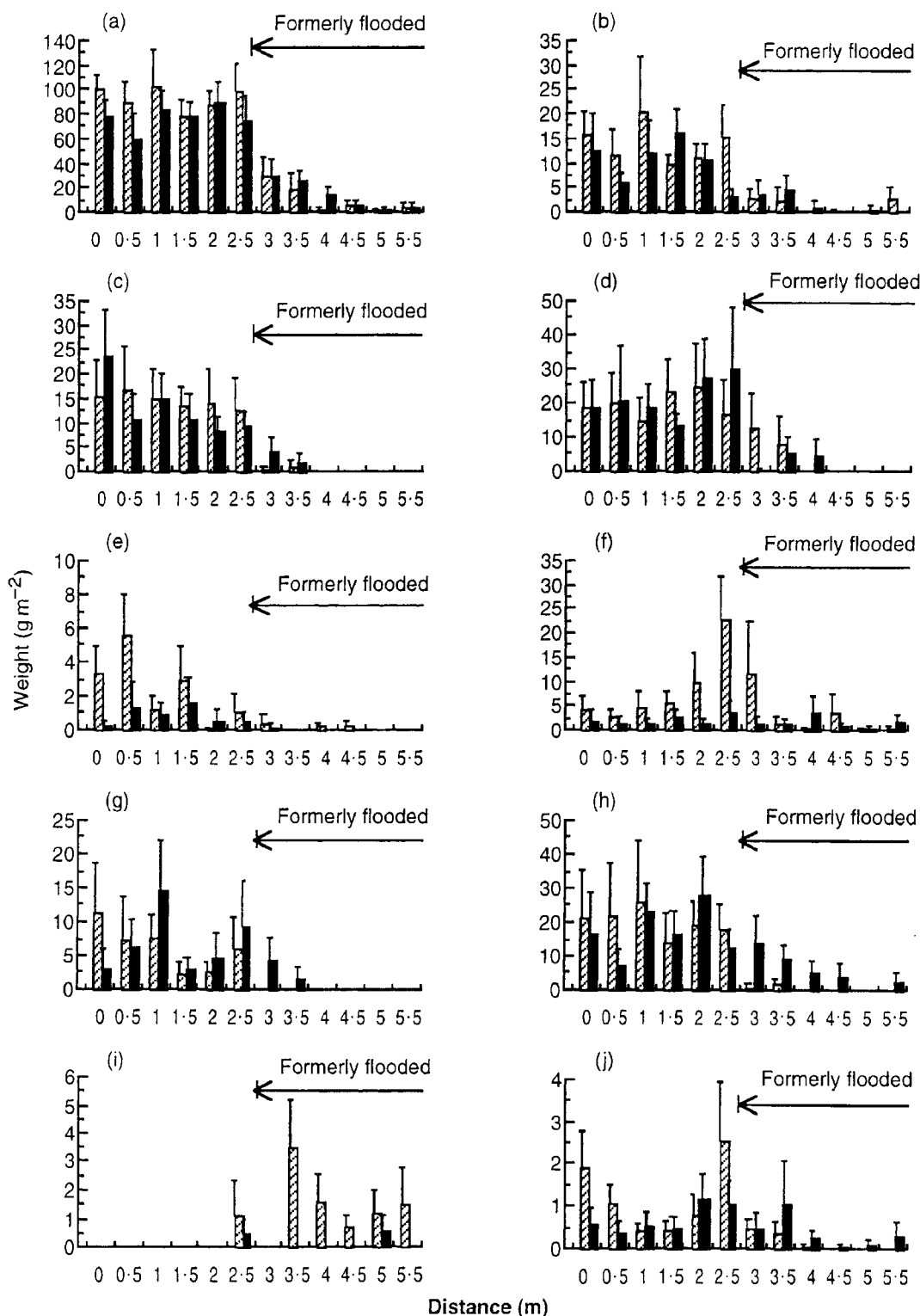
Fig. 7. Mean soil macroinvertebrate biomass in the upper 10 cm of soil along six transects running perpendicularly across the margins of unflooded grassland in March (hatched) and May (black). (a) All taxa; (b) *Allolobophora chlorotica*; (c) *Aporrectodea caliginosa*; (d) *Aporrectodea longa*; (e) *Aporrectodea rosea*; (f) *Eiseniella tetraedra*; (g) *Lumbricus castaneus*; (h) *Tipula* sp.(p.) (excluding *T. pierrei*); (i) main aquatic immature Diptera (Chironomidae larvae, *Tipula pierrei* larvae and Eristaline larvae and pupae; (j) other immature Diptera. Bars show ± 1 SE.

were capable of surviving for long periods in flooded soils if forced to remain in them. Earthworms are thought to avoid flooded soils because of their lack of oxygen and, once anoxic conditions have developed, the presence of noxious gases such as hydrogen sulphide (Mather & Christensen 1988). The extremely low soil macroinvertebrate biomass in many recently flooded parts of fields compared with most traditionally flooded grasslands could therefore have been due to the former suffering particularly severe oxygen depletion. The rate at which

Table 9. Biomass of soil macroinvertebrates in unflooded and winter-flooded grassland. Data are from six transects running perpendicularly across the margins of unflooded and winter-flooded grassland. Unfl. = unflooded half of the transect; W. fl. = winter-flooded half of the transect. Main aquatic immature Diptera refers to Chironomidae larvae, *Tipula pierrei* larvae and Eristaline larvae and pupae. Other immature Diptera were only found in the top 3 cm of soil and so figures for them are only given in (a). Paired *t*-tests were carried out on $\log(x + 1)$ transformed data

| Taxa | | Mean formaldehyde preserved biomass ± SE (gm^{-2}) | | | |
		March	May	t	P
(a) Upper 3 cm of the soil					
All taxa	Unfl.	384·6 ± 47·9	310·1 ± 27·8	1·378	0·2266
	W. fl.	41·3 ± 15·5	74·3 ± 14·0	−1·644	0·1610
Allolobophora chlorotica	Unfl.	59·6 ± 13·8	47·1 ± 15·2	0·995	0·3653
	W. fl.	4·2 ± 1·9	9·3 ± 4·2	−0·392	0·8514
Aporrectodea caliginosa	Unfl.	62·8 ± 21·4	36·9 ± 16·3	1·744	0·1416
	W. fl.	1·8 ± 1·1	4·2 ± 2·0	−1·419	0·2150
Aporrectodea longa	Unfl.	5·7 ± 2·0	8·8 ± 1·7	−1·491	0·1962
	W. fl.	1·2 ± 1·0	3·0 ± 1·9	−1·136	0·3076
Aporrectodea rosea	Unfl.	1·5 ± 1·1	0·7 ± 0·7	1·346	0·2361
	W. fl.	0·2 ± 0·2	0·0	1·000	0·3632
Eiseniella tetraedra	Unfl.	44·9 ± 18·9	13·1 ± 5·7	4·068	0·0097
	W. fl.	16·9 ± 14·6	9·8 ± 6·5	−0·312	0·7676
Lumbricus castaneus	Unfl.	36·2 ± 20·3	38·7 ± 22·3	1·019	0·3550
	W. fl.	0·0	5·5 ± 4·6	−1·474	0·2004
Tipula sp.(p.) larvae	Unfl.	113·5 ± 59·0	98·1 ± 25·1	−0·886	0·4159
	W. fl.	8·6 ± 6·3	35·3 ± 19·3	−1·080	0·3294
Main aquatic immature	Unfl.	1·2 ± 1·2	0·5 ± 0·5	1·000	0·3632
Diptera	W. fl.	7·7 ± 3·3	0·6 ± 0·6	4·599	0·0058
Other immature	Unfl.	7·1 ± 2·4	4·2 ± 0·7	0·925	0·3976
Diptera	W. fl.	0·9 ± 0·3	2·3 ± 1·0	−1·123	0·3123
(b) Upper 10 cm of the soil					
All taxa	Unfl.	558·7 ± 61·2	461·9 ± 30·7	1·254	0·2652
	W. fl.	62·0 ± 16·8	84·3 ± 18·6	−0·842	0·4384
Allolobophora chlorotica	Unfl.	86·3 ± 13·7	60·7 ± 19·4	2·303	0·0696
	W. fl.	8·2 ± 2·9	10·2 ± 4·7	0·197	0·8514
Aporrectodea caliginosa	Unfl.	87·2 ± 24·6	96·2 ± 24·4	−0·494	0·6421
	W. fl.	1·8 ± 1·1	6·9 ± 4·0	−1·433	0·2113
Aporrectodea longa	Unfl.	19·8 ± 4·6	21·5 ± 5·4	−0·526	0·6212
	W. fl.	5·2 ± 2·4	6·0 ± 4·0	0·410	0·6985
Aporrectodea rosea	Unfl.	14·3 ± 6·9	5·3 ± 2·6	1·929	0·1117
	W. fl.	1·0 ± 1·0	0·2 ± 0·2	0·564	0·5973
Eiseniella tetraedra	Unfl.	49·7 ± 21·1	13·3 ± 5·8	4·066	0·0097
	W. fl.	17·6 ± 15·1	9·8 ± 6·5	−0·193	0·8544
Lumbricus castaneus	Unfl.	37·3 ± 21·0	41·0 ± 22·8	0·844	0·3550
	W. fl.	0·0	5·5 ± 4·6	−1·474	0·2004
Tipula sp.(p.) larvae	Unfl.	120·5 ± 65·7	104·6 ± 27·5	−0·959	0·3816
	W. fl.	8·6 ± 6·3	38·1 ± 21·8	−1·107	0·3816
Main aquatic immature	Unfl.	1·2 ± 1·2	0·5 ± 0·5	1·000	0·3632
Diptera	W. fl.	8·2 ± 3·2	0·6 ± 06	4·581	0·0059

soil oxygen is depleted depends on temperature, the availability of organic matter for microbial respiration, and sometimes on the chemical oxygen demand of reductants in the soil, such as ferrous iron (Gambrell & Patrick 1978). The recently flooded grasslands tended to be flooded for shorter durations than the traditionally flooded grasslands, and in particular for shorter periods during spring and autumn (when temperatures are higher than in winter). Therefore, duration of flooding and soil temperature are unlikely to be the cause of observed differences in biomass. Recently flooded grasslands may have contained a greater volume of organic matter than traditionally flooded grasslands.

Large 'flushes' of nutrients are often released from decaying vegetation when areas are first flooded (Andersson 1982; Danell & Sjoberg 1982). It was noticeable that many sites that had been flooded for the first time in recent years had a thick, black, anoxic litter layer consisting of dead flooding-intolerant grass species such as *Lolium perenne* (L.), while traditionally flooded grasslands had no noticeable litter layer. It might be possible to prevent anoxic conditions from developing by removing most of the vegetation prior to flooding (by grazing or mowing), or by allowing water levels to fluctuate so that oxygen supply is replenished during periods of drying out.

Table 10. Mean percentage of soil macroinvertebrate biomass in the upper 3 cm of the top 10 cm of soil in unflooded and winter-flooded grassland. Data are from six transects running perpendicularly across the margins of unflooded and winter-flooded grassland. For total macroinvertebrate biomass figures are shown separately for percentages in unflooded (Unfl.) and winter-flooded (W. fl.) halves of the transects. Paired *t*-tests were carried out on arcsine transformed data and only on taxa found along five or more transects in both March and May. Main aquatic immature Diptera refers to Chironomidae larvae, *Tipula pierrei* larvae and Eristaline larvae and pupae

| | Mean percentage of biomass in the upper 3 cm of the top 10 cm of soil ± SE (gm^{-2}) | | | |
Taxa	March	May	*t*	*P*
All taxa				
Unfl.	69·5 ± 4·9	67·5 ± 4·7	0·375	0·7233
W. fl.	66·3 ± 10·8	92·4 ± 4·9	−1·962	0·1070
Allolobophora chlorotica	66·0 ± 7·2	79·7 ± 6·4	−1·857	0·1225
Aporrectodea caliginosa	74·6 ± 11·4	38·6 ± 12·2	2·310	0·0820
Aporrectodea longa	24·9 ± 7·6	64·1 ± 10·5	−2·798	0·0489
Aporrectodea rosea	16·1 ± 13·9	9·6 ± 9·6	–	–
Eiseniella tetraedra	93·1 ± 2·1	99·6 ± 0·4	−3·142	0·0348
Lumbricus castaneus	98·7 ± 0·8	88·9 ± 7·8	–	–
Tipula sp.(p.) larvae	98·5 ± 1·5	95·4 ± 2·2	1·520	0·1890
Main aquatic immature Diptera	93·0 ± 7·0	100·0	–	–
Other immature Diptera	100·0	100·0	–	–

Another possible difference between recently flooded and traditionally flooded grasslands was that most of the former were on minerals soils, while the majority of the latter were on peat. It is possible that differences in soil type could at least partially explain differences in abundance of soil macroinvertebrates in flooded grasslands, and this might repay further investigation.

The results showed that earthworms will try to vacate soil as it becomes flooded. The ratio of soil macroinvertebrate biomass in flooded compared with unflooded soil in the experiment (1 : 11·9) was similar to that found in flooded and unflooded parts of fields (1 : 10·2). Thus movement of earthworms out of flooded soils could explain the reduction in biomass found in winter-flooded areas. Earthworms are capable of moving long distances above ground. Mather & Christensen (1992) found that *Allolobophora chlorotica*, *Aporrectodea caliginosa*, *Aporrectodea longa*, *Aporrectodea rosea* and *Lumbricus terrestris* routinely made overland forays at night even under normal conditions, and Darwin (1881) recorded overland forays of earthworms of up to 13 m. In many situations, earthworms are likely to be able to vacate grassland as quickly as it floods.

Earthworms displaced by flooding will presumably initially become concentrated in the upper soil of adjacent unflooded grassland. Such concentrations of earthworms would explain the well-known attraction of soil macroinvertebrate-feeding birds, particularly lapwings, golden plovers *Pluvialis apricaria* (L.), gulls Laridae, starlings *Sturnus vulgaris* (L.) and thrushes Turdidae, to the margins of flooded grassland. Concentrations of earthworms in adjacent unflooded grassland may be significantly reduced by birds such as wintering lapwings and golden plovers (Bengtson *et al*. 1976; Barnard & Thompson 1985). The green morph of *Allolobophora chlorotica* has been found to display a cryptic advantage over its unpigmented pink morph (Satchell 1967a), and this would presumably confer a selective advantage over most other earthworm species during periods of bird predation. *Lumbricus castaneus*, which was the only earthworm species to leave the soil regularly and move through the floodwater above it, would be highly vulnerable to predation by waterbirds such as gulls, grey herons *Ardea cinerea* (L.) shoveler *Anas clypeata* (L.) and mallard *Anas platyrhynchos* (L.) in natural flood conditions.

The flooding experiment showed that many overwintering arthropods would emerge following the onset of winter flooding. These would probably accumulate on unsubmerged vegetation and also be blown to the margins of flooded areas. Being relatively cold, and therefore slow-moving, they would be easily preyed upon by insectivorous birds, especially meadow pipits *Anthus pratensis* (L.) and pied wagtails *Motacilla alba* (L.).

There was no significant recolonization of winter-flooded grassland by soil macroinvertebrates during the spring. This is not surprising, given the slow colonization rates of grassland (2·5–10 m year^{-1}) by species such as *Aporrectodea caliginosa* and *Allolobophora chlorotica* (Rhee 1969; Hoogerkamp, Rogaar & Eijsackers 1983). Recolonization of winter-flooded grassland might have been inhibited by adverse changes to the habitat there caused by winter flooding. Flooding can result in soil compaction, consolidation and loss of soil structure, which will impair air and water movement through the soil and impede earthworm movement (Piearce 1984). Recovery of soil structure is largely brought about by the actions of earthworms themselves (Satchell 1967b).

IMPLICATIONS FOR MANAGING LOWLAND WET GRASSLAND FOR BREEDING WADING BIRDS

This study has shown that winter flooding greatly reduces the soil macroinvertebrate prey of breeding wading birds, but that many of the highest densities of breeding wading birds are found on sites with only low densities of soil macroinvertebrates.

The results of the experiments suggest that the main reason for the decreased biomass of soil macroinvertebrates in flooded soils is simply due to them vacating flooded soils. However, once the majority of earthworms have sought refuge in unflooded soil, prolonging the duration of flooding is unlikely to greatly further reduce earthworm biomass.

For snipe, the overall effects of flooding will be to decrease their soil macroinvertebrate prey. Snipe are thought to require soft soil in which to probe for macroinvertebrates (Green 1988; Green, Hirons & Cresswell 1990). Flooding may in some cases increase the physical availability of the prey to snipe by helping keep the upper soil moist and therefore soft enough for them to probe. However, the wettest and therefore softest soil is likely to be that most recently uncovered by the retreating flood water. As this study has shown, such areas will only have very low soil macroinvertebrate biomass. An alternative way of providing suitable feeding conditions for snipe is to keep the upper soil moist and soft enough for snipe to probe for macroinvertebrates without flooding it beforehand. The ability to do this varies with the hydraulic conductivity of the soil. Undamaged peats tend to have high hydraulic conductivity. On undamaged peats it is possible to keep the upper soil moist and soft through lateral movement of water through the soil from surrounding water-filled ditches, particularly if these ditches are closely spaced (Silsoe College 1989; Youngs, Leeds-Harrison & Chapman 1989; Armstrong 1993). Clays usually have lower hydraulic conductivity, and it is rarely possible to keep the upper soil soft by maintaining high water levels in surrounding ditches (Armstrong 1993).

Flooding in winter and early spring is likely to improve the physical conditions for feeding lapwings and redshank. Lapwings strongly select short vegetation, usually less than 15 cm high (Klomp 1954; Lister 1964; Redfern 1982; Galbraith 1988; Ausden 1996a), and redshank also select shorter swards (Ausden 1996a). Flooding agriculturally improved grassland during winter decreases vegetation height the following spring, largely through the replacement of vigorous agriculturally improved swards with *Agrostis stolonifera*-dominated inundation grassland (National Vegetation Classification community MG13, *Agrostis stolonifera–Alopecurus geniculatus* grassland; Rodwell 1992). The height of this MG13 grassland during late spring and early summer is itself negatively correlated with the duration of spring flooding (Ausden 1996a). Therefore, the shortest, most open, vegetation is most likely to be that recently uncovered by the retreating flood water. Such areas will only have a very low biomass of soil macroinvertebrates. Lapwings and redshank also feed on aquatic Diptera larvae (Ausden 1996a; Johansson & Blomqvist 1996), and winter-flooded grassland will provide an additional food source for them in the form of Chironomidae and other aquatic immature Diptera while flood water is still present.

An alternative way of providing suitably short vegetation for feeding lapwings and redshank is through heavy grazing or by reducing soil fertility. Heavy grazing has the disadvantage that stock trample the nests of wading birds and other ground-nesting species (Beintema & Muskens 1987; Green 1988). In the Netherlands trampling of lapwing and black-tailed godwit nests is reduced using nest protectors (Guldemond, Parmentier & Visbeen 1993), although these cannot be used effectively to protect the more cryptic nests of other species such as redshank, snipe and ground-nesting passerines. Cessation of fertilizer use can reduce sward productivity within a few years, although the former unimproved vegetation (which is usually structurally more open) can take very much longer to re-establish (Berendse *et al.* 1992; Mountford *et al.* 1994).

In conclusion, the best feeding conditions for breeding snipe are likely to be provided by maintaining a high water table on peat soils without flooding them beforehand. The best feeding conditions for breeding lapwings and redshank will probably be provided by creating a mosaic of unflooded grassland, winter-flooded grassland and shallow pools on peat, clay or other soils. Unflooded grassland can provide a high biomass of soil macroinvertebrates beneath short vegetation in early spring. Winter-flooded grassland can provide short, open, conditions for lapwings and redshank to feed in during the latter part of the breeding season when the vegetation has become too tall for them to forage in elsewhere. However, winter-flooded grassland will only contain low biomasses of soil macroinvertebrates. Shallow pools will provide an alternative source of aquatic invertebrate prey. Such a mosaic of hydrological conditions is also likely to benefit a range of other wetland biodiversity.

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GULLS AND PLOVERS

The Ecology and Behaviour of Mixed-Species Feeding Groups

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than golden plovers. This difference between the two species will become important later. First, we must consider the factors likely to account for the accumulation of foraging birds on old rather than young pastures.

Pasture age and prey availability. One obvious factor which might correlate with pasture age and bias the distribution of birds is prey availability. Plovers in the study take almost exclusively earthworms during the winter (see Chapter 2) and several studies have shown positive correlations between pasture age and earthworm density (e.g. Heppleston 1971a, Waite 1981, 1983, and see Evans and Guild 1948, Guild 1951, MacFadyen 1962). Of these, however, only Heppleston (1971a) has attempted to rank fields in terms of worm density and relate this to site preference in feeding birds (oystercatchers).

We sampled the earthworm community in pasture fields using two methods: (a) randomly-placed 0.25m-square quadrats of turf and (b) 10.2cm-diameter core samples, both to a depth of 3.0cm. Both turf and core samples were hand-sorted for invertebrates and all items found were preserved in a 10% Biofix solution for later analysis. Hand-sorting was used because it is the most effective sampling technique for surface-dwelling earthworm species (Edwards and Lofty 1977). From bill-length measurements, we judged 3.0cm to be the maximum depth to which plovers could penetrate the soil and therefore the vertical limit of worm availability. Other studies (e.g. Satchell 1971, Brown 1983) have used chemical (formalin, potassium permanganate, Biofix) extraction techniques, which result in an overestimate of deep-dwelling species such as *Lumbricus terrestris* that are forced up, and an underestimate of surface-dwellers such as *Allolobophora chlorotica*, *A. caliginosa* and *A. rosea* that tend to move down. The core samples were taken to provide estimates of the vertical distribution of earthworm density, size and species composition in different fields. An important point, however, is that no bias was apparently incurred by worms burrowing away during sampling. Owing to the heavy clay content of the soil and the relatively low temperatures which prevailed during the study periods, worms moved only very slowly through the soil. In some cases, worms were still present on the cut soil surface up to five minutes after a turf sample was removed. Using the worm samples, we compared four aspects of worm availability in young and old pasture and related them to the distribution of foraging birds.

(a) Worm density. Forty-three 0.25m-square turf and 84 core samples from a total of ten fields were examined on two consecutive days in 1980/1981 when the ambient temperature was between 8°C and 10°C and weather conditions more or less constant. Within the turf samples, there was a significant positive relationship between worm density and pasture age (up to 25 years) (Figure 3.2a). The most likely reason is that, as pastures mature, there is a gradual accumulation of the dead organic matter on which worms feed (e.g.

Nordstrom and Rundgren 1974). Furthermore, old pastures are grazed by cattle for a greater part of the year and therefore have a higher organic input from dung. Worm density, however, appears to drop in very old pasture ($>$100 years) (Figure 3.2a). A possible explanation is that the thick root layer in very old pastures favours large numbers of nematodes, which reduce the amount of oxygen available to earthworms (D.B.A.T. unpubl). In addition, the largest of the earthworm species, *L. terrestris*, is found in old pasture (Edwards and Lofty 1972, Brown 1983), where it can be predatory on other species. Perhaps not surprisingly, therefore, the wet weight biomass of worms increases linearly with pasture age (Figure 3.2b) owing to the increasing predominance of larger species such as *L. terrestris* and *A. longa* (see below). To see whether ploughing itself reduces worm density in any given erstwhile pasture field, we took a number of turf samples from selected fields at the same time (between 5 and 12 March when the ambient temperature was between 8.5°C and 10°C) in three successive years. Figure 3.3a,b shows worm density in two of the fields which were undisturbed in the first two years they were sampled, but ploughed up in the third. Samples taken two months after ploughing show a significant decline in worm density. In another field which was not ploughed, worm density continued to increase in the third year, as expected from Figure 3.2a. There is, therefore, a clear positive relationship between worm density in the surface-soil layer available to plovers and the length of time since a field was last ploughed.

If the abscissa in Figure 3.1a,b is now converted from pasture age to the surface 3.0cm worm density recorded in each field, significant positive relationships for both I_c and I_n emerge. This is reinforced by stepwise partial regression analysis which examines the independent effects of pasture age and worm density on I_c and I_n. In this and all other partial regression analyses referred to, we used the stepwise forward inclusion technique detailed by Nie *et al.* (1975). Data were checked for violations of the assumptions (of normality, homogeneity of variance, linearity of relationships and weak correlation between independent variables) underlying partial regression analysis (see Pedhazur 1982). Where necessary, the frequency distribution of variables was normalised using one of several transformations (e.g. $\log_{10}$ or natural log for ratio variables, arcsin for proportions and percentage variables where their distributions approached 0 and 1 (or 100) and reciprocation where the data were highly skewed: see Sokal and Rohlf 1981 for details). When transformation still failed to satisfy requirements for the analysis, data were analysed using non-parametric methods. Independent variables yielding non-significant F-ratios were not included in partial regression analysis. For the application of partial and multiple regression techniques to other charadriiform data see e.g. Bryant (1979), Goss-Custard *et al.* (1981, 1984), Pienkowski (1983b), Ens and Goss-Custard (1984).

Table 3.5 shows the results of partial regression analysis taking into account both pasture age and mean worm density. In addition, the analysis
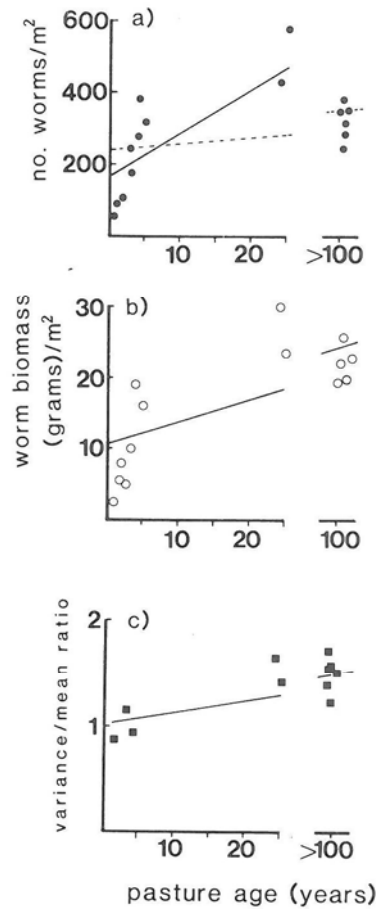
Figure 3.2: Effects of pasture age on (a) worm density, F = 14.9, p < .001 (pastures up to 25 years, solid line); the relationship is not significant when all pastures are included (broken line); (b) worm (wet weight) biomass, F = 8.2, p < .05; (c) variance: mean ratio of worm density, F = 12.96, p < .01. Data for 16 pastures

took into account field area and the number of heterospecifics present in the field; as we shall see in later chapters, both these factors have profound effects on foraging behaviour and species flock/subflock size. Table 3.5a shows that, when other factors are taken into account, the effect of pasture age on I_c for golden plovers disappears and variation in I_c is best explained in terms of variation in worm density and field area. Birds occur most regularly in large fields where worm density is high. Worm density also accounts for a significant amount of variation in I_c in lapwings, but here pasture age still exerts a significant independent effect and there is no effect of field size. When I_n is
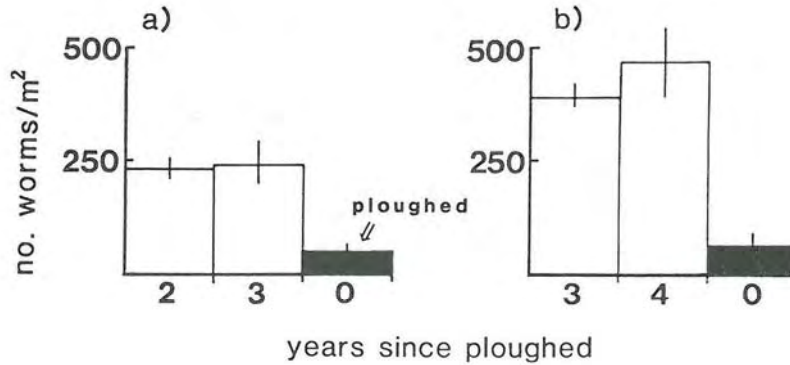
Figure 3.3: Effects of ploughing on worm density. The figure shows the relationship between mean worm density and time since ploughing in two pastures. Shaded columns show densities immediately after ploughing (see text). Bars represent standard errors

examined (Table 3.5b), the effect of pasture age disappears in both species with variation in lapwing number being explicable entirely in terms of worm density. Two important points to note here, however, are (a) I_n for golden plovers depends most on I_n for lapwings, with significant effects on field size and worm density, and (b) variation in I_c and I_n for gulls is best explained by I_c and I_n for golden plovers with no significant effect on other variables. From Table 3.5, therefore, it seems that lapwings tend to choose feeding sites partly on the basis of worm density and partly on some other correlate of pasture age, that golden plovers choose largely on the basis of intensity of use by lapwings, and that preferences in gulls are very closely correlated with those of golden plovers.

The apparent dependence of golden plovers on the distribution of lapwings suggests that they may use lapwing flocks to indicate the best fields in which to feed (see e.g. Krebs 1974, Neuchterlein 1981, Burger 1984 for similar evidence from other species). If they do, their distribution across lapwing flocks should be non-random. This appears to be the case. Significantly fewer lapwing flocks (37.1%) contain golden plovers than expected if the latter distribute themselves indiscriminately across flocks (64.6%, X^2 test comparing observed and expected, $p < .01$, n = 569 lapwing flocks). Golden plovers thus appear to be selective in their choice of flock. Indeed, we have observed them flying over four or more lapwing flocks before alighting. What criteria might birds use in choosing a flock? We recorded the size and density (no. birds/ha, see Chapter 6) of lapwing flocks joined or passed over by golden plovers. In each case we also measured the density of worms where lapwings were feeding. The results, summarised in Table 3.6, suggest that golden plovers are attracted to fields containing large flocks of lapwings and high worm densities. This is good evidence that the presence of lapwings acts as a guide to rich feeding areas.

Table 3.5: Beta values from stepwise partial regression analysis of the relationship between field characteristics and (I_c), and (I_n) (see text) in plovers and gulls

Dependent variables	Field age	Worm density	Field size	I_c or I_n L	GP	%
				Independent variables		
(a) I_c for:						
Lapwings	.52***	.40*	ns	—	—	53
Golden plovers	ns	.79***	.73***	ns	—	62
Gulls	ns	ns	ns	ns	.43***	77
(b) I_n for:						
Lapwings	ns	.87***	ns	—	—	76
Golden plovers	ns	.47***	.54**	.73***	ns	69
Gulls	ns	ns	ns	ns	.69*	96

* $p < .05$, ** $p < .01$, *** $p < .001$; significance levels for F-ratio associated with beta value. ns not significant, — variable not included. % gives % variance explained.

Table 3.6: Factors affecting the attractiveness of fields to golden plovers

Mean ± se	Fields with lapwings which: attracted golden plovers	did not attract golden plovers	Mann-Whitney U test
Worms/m^2	143 ± 14.7	119 ± 15.8	7 ($p = .047$)
Lapwing flock size	24.2 ± 4.18	11.5 ± 3.58	6 ($p = .036$)
Lapwing flock density (no./ha)	4.4 ± 0.80	3.3 ± 0.92	10 ($p = .120$)

(b) Worm patchiness. A second aspect of prey populations which may affect predator capture rate and feeding-site preference is spatial distribution. Most natural food supplies are clumped or 'patchily' distributed rather than being scattered randomly or uniformly through the environment (e.g. Taylor 1961). There is also evidence that some predators forage more efficiently on patchy food supplies (Krebs 1979). A positive correlation with the degree of worm patchiness might help to explain the remaining effect of pasture age on feeding-site selection in lapwings. We therefore took a further series of random turf and core samples, the number of samples taken in each field being the number at which the variance in worm density levelled off (e.g.

CLIMATE AND DRAINAGE

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August and September. These data are given for the area mean (in bold type) and for a selection of annual rainfalls within the area range (see above).

Sampling procedure

Technical Bulletin No. 24, *The Significance of Winter Rainfall over Farmland in England and Wales*, described the way in which a soil moisture balance sheet could be prepared on a monthly basis using observations of rainfall and calculations of potential transpiration. This involved the use of a water extraction model for the estimation of actual transpiration, which implied that most farm crops could extract a maximum of 125 mm (5 in.) from farmland soil, and that of this amount the first 50 mm was extracted at the full potential rate, the next 50 mm at half potential and the final 25 mm at quarter potential rate.

This model does not seem to be greatly at fault when compared to the results of careful experiments under grass swards. Furthermore, the results obtained by its use give estimates of return to capacity dates in the autumn which compare favourably with observations of the start of running of drains. Finally, no autumn flooding has occurred before such a theoretical return date has been reached. Experience so far therefore suggests that these calculations can be assumed to attain an acceptable standard of accuracy on a field scale, for crops with a root range similar to permanent grass.

With regard to cereal crops, the position is more complicated, as complete green cover is rarely attained until late spring, after which the plant attains a vertical height which tends to increase transpiration, it then turns from green to yellow which changes the heat balance, and finally it is harvested in late summer leaving a field which spends some time in a fallow, uncropped condition. Cereal roots, and also those of sugar beet, may also penetrate deeper than grass and there is evidence to suggest that the maximum possible extraction of water from the soil is some 20 per cent higher than that implied in the (125 mm maximum) model. This difference only becomes of importance when a dry winter follows a dry sunny summer.

Monthly soil moisture balances were calculated for almost 100 stations in England and Wales for each year in the 30-year period 1941–70. Some of these results, in terms of estimated return to capacity date, and the excess winter rainfall (which has to be drained away) between that date and the end of March, were included in the Appendices III–XIV in Technical Bulletin 24 (up to 1959/60), and the remainder now appear in the Appendix to this Bulletin.

From these 3,000 sets of data it was possible to establish formulae linking the required drainage climate parameters with known variables of rainfall and transpiration for the various major districts of England and Wales. From these formulae, data could be calculated for each agroclimatic area.

The parameters are expressed in terms of median (less on half the occasions, more on the other half), lower (or earlier) quartile (less on a quarter of the occasions) and higher (or later) quartile (more on a quarter of the occasions). It must be stressed that the median is not the average unless the distribution of the parameter about the average is symmetrical, which rarely occurs in rainfall statistics. Because a small number of years with heavy rainfalls are counterbalanced by a larger number of years with low rainfalls, the median is generally a lower value than the average. Rainfall has what is known as a skew distribution. The median

and the 50 per cent range (quartile to quartile) are chosen as the most useful for planning purposes.

Excess winter rains

The Lower Quartile, Median, and Higher Quartile are given for selected points in the range of average annual rainfall totals within each area, together with the areal mean. These data have been quoted to the nearest 5 mm with the equivalent in inches, although in some of the upland wetter areas, the figure has been rounded off in 25 mm stages. They are probably correct within 5 per cent, with the proviso that they could be up to 25 mm too high in fields under continual cereal cropping.

The greatest errors probably occur in rough grazing areas where insufficient is known of the depth of soil and hence the maximum soil moisture deficit in summer. Surface run-off is also an unknown factor in uplands with pronounced slopes. Although precision cannot be attained, the general order of magnitude is certainly reliable.

Return to field capacity

In the yearly month by month calculations, return dates were expressed in 10-day intervals. The 30-year median and quartile data are presented as individual dates, with a possible error of plus or minus 3–4 days. Again some allowance must be made for the effect of late harvested cereals or sugar beet, and the return dates following such crops are likely to be some 7–10 days later in early autumn, and possibly 15–20 days later in mid-winter. In other words, the difference grass-cereals is small in time if the autumn is wet, but if autumn and winter are dry the return date may be considerably delayed following the deeper rooted crops, as an extra 25 mm of rain is needed to replenish the soil. Summer fallowing will advance the return date.

In high rainfall areas it is often impossible to identify the dates on which the drains stop running and when they recommence. In such cases no estimate has been attempted, and the symbol (*) has been used signifying 'indefinable'. This implies that the soil is rarely below field capacity for any appreciable continuous period.

End of field capacity

Although soil moisture balance models can produce a good estimate of the return to field capacity in the autumn, it is far more difficult to specify or to estimate the end of capacity in spring, mainly because it is often an ill-defined intermittent process. The results quoted are the best that can be estimated on the basis of past weather, but it must be stressed that the April rainfall and the April transpiration are very nearly equal in most farm areas. Therefore, any small change in either parameter will have a major effect on the dating of the end of capacity. Thus although dates in March or May can be relied upon to some extent, the April figures could change by some 10–15 days following a very small modification of climatic conditions.

Areas 12 and 13

		Area 12 Mean				Area 13 Mean			
Annual rainfall	mm	540	643	700	780	560	655	725	800
	in.	21·3	25·3	27·6	30·7	22·0	25·8	28·5	31·5
Excess winter rain									
Lower Quartile	mm	25	105	155	225	40	110	175	245
MEDIAN	mm	100	195	250	325	120	200	270	345
Higher Quartile	mm	200	305	360	440	230	315	385	460
Return to field capacity									
Earlier Quartile		Oct 26	Oct 3	Sep 18	Aug 28	Oct 21	Oct 2	Sep 14	Aug 23
MEDIAN		Dec 10	Nov 13	Oct 26	Oct 4	Dec 5	Nov 11	Oct 20	Sep 28
Later Quartile		Jan 11	Dec 15	Nov 30	Nov 9	Jan 6	Dec 14	Nov 24	Nov 3
End of field capacity									
Earlier Quartile		Mar 10	Mar 17	Mar 24	Apr 1	Mar 13	Mar 19	Mar 27	Apr 6
MEDIAN		Mar 29	Apr 8	Apr 16	Apr 26	Apr 2	Apr 11	Apr 20	May 1
Later Quartile		Apr 23	May 4	May 11	May 20	Apr 25	May 6	May 14	May 24
Heaviest rainfall									
Expected in 1 day									
in 1 year	mm	15	19	21	24	15	18	21	24
in 2 years	mm	19	23	26	30	18	21	25	28
in 10 years	mm	26	31	34	39	24	27	32	35
Expected in 5 days									
in 1 year	mm	28	36	42	50	26	33	40	46
in 2 years	mm	34	43	49	57	31	40	47	55
in 10 years	mm	45	55	62	72	41	51	59	67
Soil moisture deficit									
End June									
Lower Quartile	mm	65	53	44	32	62	51	40	29
MEDIAN	mm	75	68	64	58	73	67	62	56
Higher Quartile	mm	92	86	81	76	90	85	79	74
End July									
Lower Quartile	mm	77	63	52	39	73	61	49	36
MEDIAN	mm	95	85	76	66	93	84	74	64
Higher Quartile	mm	112	103	96	88	110	102	94	86
End August									
Lower Quartile	mm	67	40	20	0	61	37	15	0
MEDIAN	mm	90	75	61	44	87	74	57	40
Higher Quartile	mm	118	105	93	79	115	105	90	75
End September									
Lower Quartile	mm	51	10	0	0	50	6	0	0
MEDIAN	mm	79	60	44	15	75	61	42	0
Higher Quartile	mm	127	105	89	69	123	105	85	65

Old Little Humber Farm

Wet Grassland Creation

Soil and Hydrological Survey

For

Able UK Ltd

Contents

5.   Climate

5.1   Methodology

5.1.1   Climate data was consulted to gain an understanding of the typical climate in the area surrounding the Old Little Humber Farm Compensation Site. The site falls within Agro-Climatic Area 13 for Humberside taken from Smith and Trafford (1976).

5.1.2   This data was used to calculate the balance of rainfall over transpiration and to produce a water balance graph demonstrating the balance of rainfall and potential transpiration at the site over the period of one year.

5.2   Results

5.2.1   Graph 1 below shows a typical water balance curve for the site. This balance assumes that there is some loss of water from the site and shows in general there will be a cumulative deficit of water from mid-April until early November.

5.2.2   However it also shows there to be a positive balance of rainfall over potential transpiration from mid-August until April. The aim must therefore be to manage the water in such a way that the periods of excess can cover the period's deficit giving a balance suitable for the site.

Graph 1: Water Balance, March
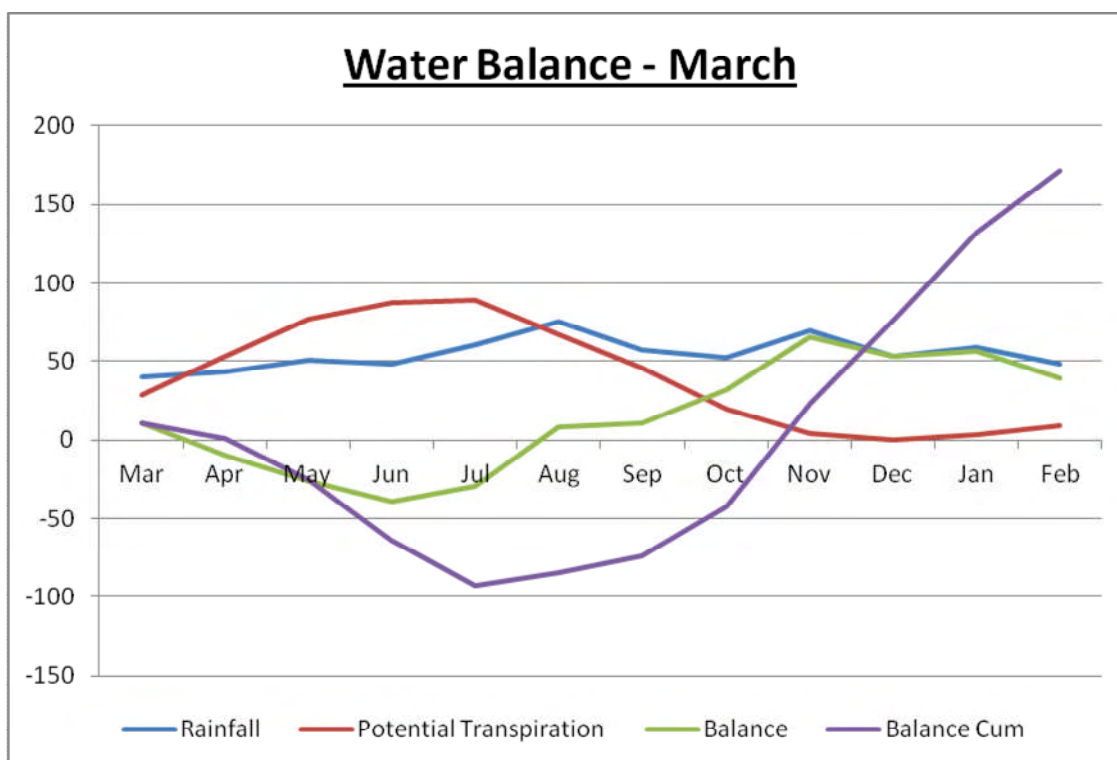
5.2.3   From the data presented in the above graph and the climate and drainage information from
        Smith and Trafford (1976) the likely water status through the year given the rainfall evaporation
        and drainage status at present is shown in Box 1 on the following page.

Box 1: Water Balance Results

The land will return to field capacity i.e. be wet enough to begin to produce drainage;

- Early quartile (wettest year) - October
- Later quartile (driest year) - December
- Median - November

From this time onwards the ground will be saturated and begin to produce drainage. The excess water (given as a depth) will be (assuming the median case);

- End November 50mm
- End December 100 mm
- End January 150 mm
- End February 170 mm
- End March 10mm

After this period the ground will dry below field capacity and the soil will go into moisture stress. Taking the three examples again this will happen;

- Early quartile (driest year) - Mid March
- Later quartile (wettest year) - Mid May
- Median - Early April

From this time the ground will come under greater moisture stress as follows;

June

- Driest quartile - 85mm
- Wettest quartile – 50mm
- Median – 67mm

July

- Driest quartile - 102mm
- Wettest quartile – 60mm
- Median – 85mm

August

- Driest quartile - 105mm
- Wettest quartile – 37mm
- Median – 74mm

> *September*
>
> - Driest quartile - 102mm
>
> - Wettest quartile – 6mm
>
> - Median – 61mm
>
> *October*
>
> - Driest quartile - 75 mm
>
> - Wettest quartile above field capacity
>
> - Median – 25 mm
>
> The land now returns to field capacity as shown at the beginning of the cycle. What is therefore evident is that the ground moisture status at present is not acceptable for the habitat conditions required.

5.2.4   Clearly this shows that taking the median case from mid-May until the end of October the ground will be under various levels of moisture stress. This stress will be longer in a dry year and less in a wet one.

The Wet Grassland Guide

Managing floodplain and coastal wet grasslands for wildlife

Institute of Terrestrial Ecology

ENGLISH NATURE

RSPB

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Appendices

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3.2.5 Agricultural management practices

A number of techniques have evolved to boost the agricultural productivity of grasslands, namely the application of:

■ inorganic fertiliser

■ farmyard manure and other organic fertiliser

■ lime

■ herbicides.

All the above are damaging to grasslands of high botanical interest and their effects are considered below.

Inorganic fertilisers

For grasslands generally, there is ample evidence that the use of nitrogen fertiliser or compounds of nitrogen, potassium and phosphorous (NPK) leads to a reduction in species richness and diversity. For wet grasslands in particular, recent detailed studies at Tadham Moor in Somerset have shown that even very low rates of nitrogen application (in agricultural terms – eg 25 kg N/ha/yr) can significantly reduce species richness. Phosphorus and potassium also cause marked reductions in species richness. There is growing evidence that phosphorus is the most significant element in affecting botanical composition (Mountford *et al* 1994b).

The inescapable conclusion is that no fertilisers should be used where conservation of botanical diversity is an objective. Sites can take many years to recover from the effects of fertilisers. At Tadham, after only four years of using 25, 50, 100 and 200 kg N/ha/yr, it was predicted that 3, 5, 7 and 9 years respectively will be required for the grassland to return to a composition approaching that found without fertiliser (Mountford *et al* 1996). Longer periods are likely to be needed for grasslands with a longer history of fertiliser application.

The more uniform and lush vegetation which develops when fertilisers are used is also generally less suitable for invertebrates and most breeding waders and wildfowl. More intensive grazing or cutting to control the increased growth creates an additional problem. However, overwintering grazing geese can benefit from the effects of fertilisers in boosting sward production.

High rates of nitrogen fertiliser result in replacement of old grazed hay meadows (MG5), water meadows (MG8) and fen-meadows (M22 and M23) by dairy and fatting pastures (MG6) and improved swards (MG7). Very similar changes occur following drainage, with wet grasslands becoming replaced by more species-poor mesotrophic types. Also biological drying can be caused by fertiliser use, as the more productive vegetation has an increased evapotranspiration rate.

Farmyard manure and other organic fertilisers

There are four main categories of organic fertiliser:

■ slurry

■ farmyard manure (FYM)

■ sewage sludge

■ others (including dried blood, guano, hoof-, horn- and fish-meal, seaweed and wool shoddy).

The impact of organic fertilisers is similar to that of inorganic fertilisers. For the same reasons, their use in grasslands managed for high existing botanical interest is inadvisable. However, it is important to recognise that making hay without using FYM (or some other fertiliser) is likely to be agriculturally uneconomic. Hay cropping removes phosphate and potash from the soil; hay at 85% dry matter removes 7 kg of P_2O_5/tonne of hay and 21 kg of K_2O/tonne of hay. As buying in hay is undesirable because of the risk of importing weeds and nutrients and hay making is an intrinsic part of the traditional management that has maintained desirable floras, there may be both an economic and conservation

argument for the continued use of some FYM on some sites (Simpson and Jefferson 1996).

Where appropriate, well-rotted FYM should be used; for example, cattle dung mixed with straw that has been stored for at least four months. Small quantities are recommended, eg a single dressing of <20 t/ha every 3–5 years, provided that there is clear evidence of traditional usage. FYM should always be spread outside the breeding season where there are ground-nesting birds, and care should be taken to ensure that it is well distributed. Applications of FYM are most appropriate in flood meadow (MG4), old grazed hay meadow (MG5), and inundation grassland (MG11, MG13) communities.

More extensive guidance on the use of FYM on semi-natural (meadow) grassland can be found in Simpson and Jefferson (1996).

Lime

In some wet grasslands there is a long tradition of lime application to offset losses by leaching and cropping and to prevent excessive acidification. Acidification can reduce the diversity of plant communities. It can therefore be advisable to lime occasionally, for example every 5–10 years, on sites with a tradition of use. Dressings should not exceed 3 t/ha of calcium oxide (or equivalent) and should always be made outside the bird breeding season (ie 16 July – 14 March).

The practice of liming should be avoided in semi-natural grasslands with no clear tradition of use (see Crofts and Jefferson 1994). Liming can make phosphorus both more available to growing plants and more likely to leach into drainage channels where it may damage distinctive plant and animal communities.

Herbicides

Herbicides are used to eliminate or control undesirable plant species. Many species of conservation interest are physiologically similar to target species, and are therefore also vulnerable. Such agrochemicals should therefore be used selectively (eg by weed-wiping – see section 4.4.6) or very sparingly, if at all, on semi-natural wet grasslands.