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Can ecosystem services be integrated with conservation? A case study of breeding waders on grassland

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We review published studies to show that changes in soil moisture levels have significant impacts on a range of wading bird species that use UK lowland grassland, including wet grassland, and obtain their food predominantly by probing the soil. We examine both the hydrological and the ecological literature and assess how management options could alter (1) ecosystem services (via water quality and flooding) and (2) habitat quality for wading birds. The combination of biodiversity goals with broader ecosystem services has been widely advocated and we suggest that appropriate management at multiple scales (e.g. small-scale: ponds; large-scale: integrated washlands) could potentially provide both ecosystem services and habitat for wading grassland birds. However, there is only a limited base of evidence on which to assess the potential linkage between these two areas, particularly for non-wading bird species. Future work should be directed at identifying (1) how crop yield, ecosystem services and biodiversity relate to each other, (2) the extent of land needed to be managed to benefit these multiple purposes and bring about measurable gain (e.g. one or two ponds may make significant inroads in reducing run-off and pollution but make little difference to wading birds) and (3) solutions to the challenges of setting up management options at large spatial scales (e.g. catchments).

Keywords: agri-environment schemes, grassland management, soil moisture, UK.

It is well established that farmland bird population declines are strongly linked to agricultural intensification (e.g. Donald *et al.* 2001) and that changes in climate are also predicted to affect these bird populations (Huntley *et al.* 2007). Climate change is likely to have a significant regional impact on groundwater levels and the availability of suitable feeding habitat, through a combination of changes in seasonal rainfall and rising sea levels (Hulme *et al.* 2002). In addition, agriculture, at a global scale, faces the challenge of providing food for approximately 50% more people by 2050 (Green *et al.* 2005). There is increasing interest in ecosystem services as a means of accounting for the full range of environmental, social and economic benefits provided by land management. A multi-functional farming landscape could potentially provide food, ecosystem services (e.g. flood control) and suitable habitats for biodiversity. Given that demands on land will increase, one key issue for the conservation of biodiversity is whether it can be linked directly to the provision of wider ecosystem services for which political pressure and budgets are often greater. This idea has been proposed in the literature many times (e.g. Morris *et al.* 2004, McInnes 2007). Here, we review the literature to provide examples to test this idea using birds feeding predominantly on soil invertebrates in lowland grass fields in the UK.

We reviewed the hydrological and ecological literature to describe (1) changes in soil moisture on lowland grassland, (2) the link between soil moisture, foraging by farmland birds and their macro-invertebrate prey, (3) evidence for population-level effects of soil moisture changes on birds, and (4)

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how changes in climate, via rainfall patterns and rising sea levels, may alter grassland. We reviewed 251 papers resulting from a literature search in the ISI Web of Knowledge (up to 18 September 2009) using a combination of search terms including the keywords 'UK, grassland, breeding/wintering birds, invertebrates, flooding, pollution, legislation and wetland'. The initial aim of the review was to consider all bird species living on farmland. However, most of the relevant literature identified focused on waders. Therefore, the study focuses on this group, with reference to studies of other bird species in the discussion. Finally, we summarize how proposed management options may affect both hydrology and farmland birds that derive the majority of their invertebrate prey from the soil. We consider how these management options could be applied in two different situations: (1) protected areas in which the main focus is nature conservation and (2) the wider countryside in which the focus of land use is varied but typically focuses on agricultural yield.

MECHANISMS LEADING TO THE DECLINE IN AREA AND SUITABILITY OF LOWLAND GRASSLAND HABITAT

Effects of land drainage

The main aim of lowering the water table of wet grasslands has been to facilitate reduced water-logging in the upper layers of soil to increase the length of the grazing season (Bradbury & Kirby 2006). By the end of the 19th century, most of lowland England's wetlands (Werritty 2006) and 5 million hectares of lowland floodplains (Smout 2000) had been drained and converted into productive agricultural land. Many of these drainage systems fell into disrepair during a period of agricultural recession, following a collapse in farm prices after World War I (Dobbs & Pretty 2004). To prevent further recession (Dobbs & Pretty 2004) and promote national self-sufficiency in food production (O'Connell *et al.* 2004), the UK government encouraged intensification and modernization of British agriculture from 1930 onwards. Existing drainage systems were restored and, to increase agricultural output, the drainage of additional wetlands was encouraged through government subsidies (Acreman *et al.* 2007). In the 1970s, the practice reached a peak of around 100 000 ha/year and was particularly common in the clay-dominated arable areas of eastern England (Green 1979). In addition, complex ditch networks, which naturally divided wetlands into small fields (Thompson 2004) and maintained high water tables, were removed to create larger fields. Few of the remaining ditches retain moisture throughout the year due to under-field drainage.

On grassland, loss of botanical heterogeneity and invertebrate species-richness is often associated with improved drainage and the subsequent increased use of fertilizer, reseeding with ryegrass mixes, increased stocking densities and earlier grazing seasons (Morris 2000, Wilson *et al.* 2005). Short periods of high-intensity stocking on clay grazing marshes and the use of heavy machinery result in the formation of a hard surface mat of vegetation and compaction of the soil, leading to high surface penetration resistance (Armstrong 2000, Hamza & Anderson 2005). Increased fertilizer application is likely to (1) have a negative impact on the existing soil moisture deficit because increased availability of nitrogen for plant uptake increases plant growth and evapotranspiration (Garwood 1988) and (2) reduce plant diversity and consequently the range of invertebrate prey present in the sward (McCracken & Tallowin 2004). These reduce both abundance and accessibility of food for wading birds. In addition, nests and chicks are vulnerable to trampling by livestock (e.g. Green 1988) and the timing of stock turn-out and mowing is known to inhibit re-nesting and therefore limit breeding success (Beintema & Muskens 1987).

Widespread land drainage over the last 200 years has resulted in a reduction in the quantity of grassland through conversion of wetland habitats to arable farmland. Subsequent intensive management of the remaining grassland resource means that it is of limited quality for wading birds through reductions in suitable nesting habitat, direct effects of trampling on nests/chicks, soil degradation and compaction, and reduced abundance, availability and access to invertebrate prey.

CURRENT POLICY

Grassland systems outside areas managed specifically for other purposes (e.g. nature reserves focused on biodiversity needs) primarily produce agricultural goods (mainly grass for fodder). These agricultural grass fields could potentially yield a range of indirect benefits, including flood protection, biodiversity and water quality. Funds previously committed to support farm output are increasingly diverted to encourage land managers to deliver these environmental benefits (Defra 2002). A number of specific measures promoted by these policies are likely to become increasingly important in the conservation and protection of water resources. However, the voluntary nature of the schemes may result in poor uptake (Davey *et al.* 2010). Also, if payments are not considered sufficient, farmers will be reluctant to install measures that they may perceive to be detrimental to their livelihoods. Currently, there is little research on the trade-offs between agricultural yield, ecosystem services and biodiversity (e.g. Vickery *et al.*1994, Morris *et al.* 2008) upon which to base policy decisions and guide levels of compensation schemes.

Increasingly, government and conservation agencies recognize the benefits of a systematic approach to conservation, with clear objectives and measurable targets, and the need to integrate grassland management with wider issues relating to water management (e.g. flood mitigation) (Benstead *et al.* 1997). For example, the governmental strategy for flood risk management in England, *Making Space for Water*, emphasizes the need for integrated land and water management through ecological enhancement and non-structural solutions (e.g. wetlands) to 'reduce the threat to people and their property and deliver environmental, social and economic benefit consistent with sustainable development principles' (Defra 2005a). The ecosystem approach was first adopted by the Convention for Biological Diversity (CBD) in 1992. It provides a framework for the integrated management of land, water and living resources to achieve a number of CBD objectives, including the conservation of biodiversity and the sustainable use of its components. Since then it has been adopted across the European Union (EU) as an approach by which to deliver several environmental directives, strategies and agreements (Apitz *et al.* 2006, McInnes 2007) and to achieve sustainable development, through the maintenance of fully functioning ecosystems (Laffoley *et al.* 2004). The ecosystem services approach is now widely recognized and reflects the emphasis placed on the benefits that society can derive from ecosystems: provisioning, regulating, supporting and cultural (Millennium Ecosystem Assessment (MEA) 2005). Within the EU, the Water Framework Directive (WFD) promotes the integration of land use and water policy and the positive use of floodplains through the development of River Basin Management Programmes, by member states. To meet the ecological water standards set by the WFD, member states are required to address issues relating to sustainable water resource management in individual river basin management systems. In England, projects such as the Fens Floodplain Project, part of the EU's Wise Use of Floodplains Project, assess how floodplain wetlands contribute to water resource management and identify ways to help implement the WFD throughout the EU. Catchment Flood Management Plans have also been developed by the Environment Agency to monitor the effects of factors such as changes in land management, loss of habitat and climate change on floods at the river catchment scale, with an aim of identifying effective methods of long-term integrated flood risk management (Environment Agency 2004). Because of the need to comply with multiple objectives that may be the remit of different government and non-governmental organizations and departments, partnership working and cross-stakeholder support is central to the successful application of the ecosystem services approach (ELP 2008). To understand how best to manage grasslands to achieve different goals, such as nature conservation, maximizing agricultural yield (to maintain food security) and minimizing flood risk, research needs to address how the different competing needs relate to each other.

SOIL MOISTURE, WADING BIRDS AND THEIR PREY

Soil penetrability, wader foraging and habitat selection

A range of wading species that feed predominantly on soil invertebrates are associated with lowland grasslands during the breeding season (Common Snipe *Gallinago gallinago* L. Green 1986, Common Redshank *Tringa totanus* L. and Northern Lapwing *Vanellus vanellus* Baines 1990, Ausden *et al.* 2003, and European Golden Plover *Pluvialis apricaria* L. Pearce-Higgins & Yalden 2003). These species feed on macro-invertebrates such as earthworms and tipulid larvae. Grassland also provides foraging opportunities for a number of species during the winter, including Golden Plover and Lapwing (Fuller & Youngman 1979, Tucker 1992). Permanent pasture (grass more than 5 years old) is of particular importance because of significantly higher earthworm biomass compared with other field types (e.g. bare till, winter cereal) (Tucker 1992).

Soil surface strength is correlated with soil moisture content. For most soils, this is associated with the water table depth from the surface (Armstrong 2000). Raised water levels keep the surface soil moist, increase soil surface penetrability (Gerard 1967, Green *et al.* 2000) and reduce vegetation growth when surface water is present (Ausden *et al.* 2001). The lowering of field water levels reduces soil penetrability, making the ground too hard for surface probing. It should be noted, however, that the relationship between soil moisture and penetration resistance can differ between soil types. For example, sandy soils can have greater penetration resistance when wet than dry. Soil types can be differentiated by their hydraulic conductivity (Armstrong 1993). For instance, clay soils have low hydraulic conductivity; water will not move easily through these soils and they tend to retain surface water for long periods. Peat soils tend to have highly variable hydraulic conductivity, influenced by soil particle size, shape and structure, and degree of decomposition (Wong *et al.* 2009). There is a range of other factors, such as aspect, slope and vegetation cover, that will further alter soil moisture. However, changes in the water table are likely to result in variations in soil moisture and have been shown to have marked effects on waders. Green (1988) used a penetrometer to measure the maximum force (kgF) required to push a steel probe 10 cm into the soil, mimicking the behaviour of the beak of a Snipe and providing a measure of penetration resistance (Green 1986, 1988, Green *et al.* 1990). Soil surface penetrability is an indirect measure of soil moisture that provides an indication of the difficulty a bird might be expected to have when probing the soil to forage (Armstrong 2000).

Wet features, such as ponds, ditches, footdrains and rills (Table 1) may retain water throughout the breeding season, maintaining a higher water table in the surrounding soil than in other parts of the field. The area affected is dependent on soil type. Milsom *et al.* (2000) established that the distribution of breeding Redshank and Lapwing on coastal grazing marshes is strongly positively influenced by the availability of rills that retained water in early June. Footdrains have been successful in maintaining localized shallow surface water in spring and the density of associated ‘footdrain floods’ positively influences field selection in Lapwings (Eglington *et al.* 2008). As birds concentrate on water margins to feed, the perimeter of these wet features is more important than their area. Redshank breeding densities are positively correlated with wet feature length, the combined total of rills, footdrains and pools (Fig. 1) (Smart *et al.* 2006). The success of any scheme of wet

Table 1. Definitions of wet feature types.

	Definition
Pond	A body of water, both natural and man-made, between 1–2 m^2 and 2 ha in area, which may be permanent or seasonal (Davies *et al.* 2008a, Williams *et al.* 2008)
Paired ponds	Paired ponds, varying in size from 1.5 m^2 to approximately 50 m^2, located alongside field ditches. The upper pond is fed by water diverted from the ditch and the second pond is fed, via a vegetated buffer strip, from the first pond, before overflowing back to the ditch system (Bailey *et al.* 2007)
Ditch	Man-made channel created primarily for agricultural purposes and which usually follows linear field boundaries (Davies *et al.* 2008a)
Bunded ditch	An existing ditch which has been dammed (bunded), to retain water (Bailey *et al.* 2007)
Footdrain	A shallow channel historically used for drainage on grazing marshes (Eglington *et al.* 2008)
Footdrain flood	An area of surface flooding resulting from water spilling over from footdrains (Eglington *et al.* 2008)
Rill	Relict saltmarsh creek and drainage channels (Milsom *et al.* 2000)
Small Constructed Wetland	A wetland constructed in a terraced design to reduce downhill flow velocity by means of a series of weirs (Raisin *et al.* 1997)
Wetland	‘Areas of marsh, fen, peatland or water, whether natural or artificial, permanent or temporary, with water that is static or flowing, fresh, brackish or salt, including areas of marine water the depth of which at low tide does not exceed six metres’ (Ramsar Convention 1971)
Washland	‘An area of the floodplain that is allowed to flood or is deliberately flooded by a river or stream for flood management purposes, with potential to form a wetland habitat’ (Morris *et al.* 2004)

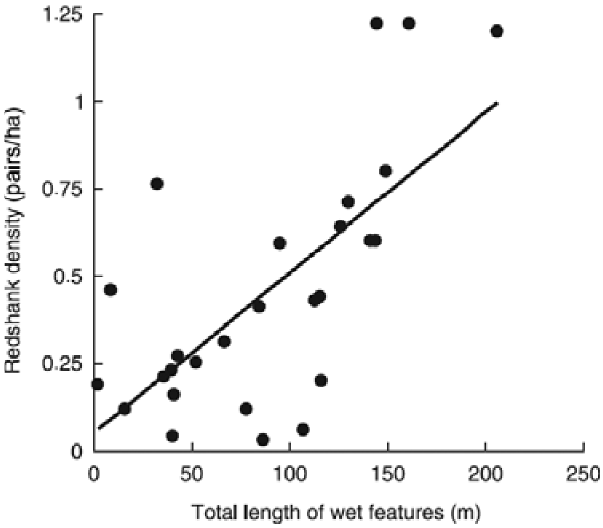

Figure 1. The relationship between Redshank breeding density and the total length of wet features (footdrains, rills, pools and ditches) in fields occupied by breeding Redshank on grazing marshes ($y = 0.005x + 0.052$, $R^2 = 0.47$, $n = 27$, $P < 0.001$) (Smart *et al.* 2006 – reproduced, with permission, from *J. Appl. Ecol.*).

feature creation is dependent on the ability to maintain wetness throughout the breeding season. In the case of breeding Snipe, drying out can lead to limited opportunities for replacement nesting after early breeding failures, as breeding ceases when the penetration resistance exceeds 5.8 kgF (Green 1988). However, when good feeding conditions persist (e.g. penetration resistance is < 5.8 kgF) Snipe will continue to initiate nests well into July, potentially doubling the number of chicks hatched (Green 1988). It is important to note two caveats: (1) above-ground prey (on which many wading species also feed) could also be an important influence on distribution and (2) soil moisture can affect prey both within and above the soil surface.

Soil invertebrates

Moist soils support larger densities of soil invertebrates than dry soils (Milsom *et al.* 2000). Moisture is one of the main factors determining earthworm abundance in the top 5–10 cm of soils (Gerard 1967, Green *et al.* 2000, Peach *et al.* 2004a) and influences pupation rates and larvae survival of terrestrial (Meats 1974) and obligate aquatic invertebrates. As the soil surface dries out, earthworms descend deeper into the soil and become unavailable to foraging birds, forcing them to switch to potentially less nutritional invertebrate prey (Gruar *et al.* 2003). Important prey, such as crane flies (e.g. *Tipula paludosa* L.), can be adversely affected by desiccation if the soil dries out quickly at a vulnerable stage in their life cycle (McCracken *et al.* 1995). The maintenance of high water tables until mid-summer is therefore important for ensuring that earthworms remain within reach of probing birds and that soil invertebrate larvae remain viable as prey.

On sites subjected to flooding, the water-holding capacity and organic matter content of soil will have an influence on invertebrate survival because prolonged waterlogging can have an adverse affect on soil-dwelling invertebrates and larvae (McCracken *et al.* 1995, Plum 2005). Tipulid larvae may die as a result of surface flooding (Meats 1970) and decaying vegetation on previously fertilized grassland can cause anoxic conditions harmful to the soil fauna (Ausden *et al.* 2001). Managed surface flooding is used to increase the area of shallow flooded grassland and soft, wet soil conditions available for breeding waders. However, prolonged surface flooding in winter and/or spring reduces the abundance of soil macro-invertebrates and can result in compaction and consolidation of the upper soil, making it difficult for birds to probe (Ausden *et al.* 2001).

When flooding is (re)introduced in grassland, initially it can attract large numbers of wading birds as prey migrate to the soil surface, but numbers decline with time as terrestrial soil invertebrates species struggle to survive in soil with prolonged flooding. Ausden *et al.* (2001) found soil macro-invertebrate densities in unflooded pasture land were 10 times higher than in flooded wet pasture land. If flooding is at a large spatial scale, re-colonization by macro-invertebrates from unflooded refuges is unlikely to occur (Plum & Filser 2005) due to the negative effect of regular flooding on spring populations of soil macrofauna. To maintain viable populations of annelids, the time interval between two flood events should not exceed the development time from cocoon to adult of the earthworm species present (approximately 6 months) and during the spring, when earthworms serve as food for ground-probing birds, a new inundation in this recovery period should be avoided or kept short (Plum & Filser 2005). A trade-off therefore exists between maintaining optimum soil penetration resistance for probing birds and the adverse effects of too much flooding

(Smart *et al.* 2008). Unflooded grassland provides a high biomass of soil macro-invertebrates beneath vegetation, whereas winter flooded grassland provides damp surface soil with short, open conditions for feeding.

Prime conditions for both invertebrate survival and reproduction and for foraging waders require a trade-off between soil conditions. Dry summer soil conditions result in the death of invertebrate larvae and force earthworms to descend deeper into the soil, thus reducing prey availability. Conversely, prolonged flooding results in invertebrate prey that are accessible but at low abundance because excessive waterlogging reduces populations. The exact requirements of different wading bird species at different times of the year are likely to differ in the precise optima of this relationship but the general themes described above are likely to hold.

CHANGES IN SOIL MOISTURE AND WADER POPULATIONS

Wet grassland breeding wader distribution is strongly related to site wetness (e.g. Green & Robins 1993, Vickery *et al.* 1997, Paillissona *et al.* 2002). In the Somerset Levels, the range contraction of breeding Snipe and Redshank accompanied the acceleration of drainage improvement that began in the late 1960s (Williams & Bowers 1987). Recent work has shown that Snipe breeding populations are more likely to have persisted in fields where the soil conditions are wet and soft (Fig. 2; Smart *et al.* 2008). Despite the introduction of management aimed at improving conditions, breeding Snipe populations have continued to decline (Ausden *et al.* 2001, Ausden & Hirons 2002, Wilson *et al.* 2004) with declines more

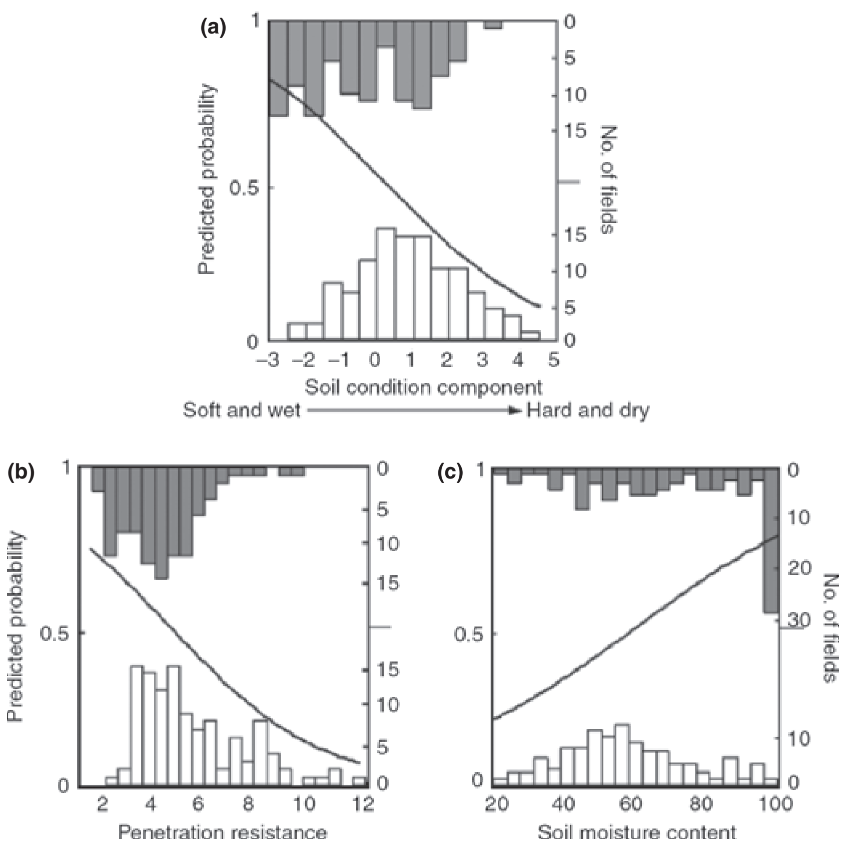

Figure 2. The probability of breeding Snipe *Gallinago gallinago* populations in fields persisting (numbers maintained, gained or increased) or becoming extinct between episode 1 (1990/1991/1992) and episode 2 (2006) in relation to (a) the soil condition component, which is the gradient from soft and wet to hard and dry soil conditions, (b) penetration resistance (mean visits 2 and 3, kgF) and (c) soil moisture content (mean visits 2 and 3, % water). Bars show the frequency distribution for fields where Snipe persisted (grey bars) and fields where Snipe became extinct (white bars). The line shows the fitted logistic regression curve (Smart *et al.* 2008 – reproduced, with permission, from *Anim. Conserv.*).

marked on mineral soils and in the south and east of England (Smart *et al.* 2008).

Wader population trends are affected by a range of factors, e.g. conversion of grassland to arable (Robinson & Sutherland 2002). However, in England, declines of wader species have been so extreme that 64% of wet grassland wader populations are currently concentrated onto eight key wet grassland sites. These declines may have been driven by changes in wetness rather than land-use change because, since 1982, there has been little further loss of grassland habitat (Wilson *et al.* 2005).

CLIMATE CHANGE AND GRASSLAND

Future UK climate change scenarios predict that, by the 2080s, annual average temperatures are likely to rise by more than 2 °C. In the southwest, summer precipitation may decrease by up to 40%, under the medium emissions scenario (Jenkins *et al.* 2009) and large parts of England may experience more than a 40% reduction in summer soil moisture (Hulme *et al.* 2002). It is likely that the UK will experience 'higher water demand, more widespread water stress with increased risk of drought, more water quality problems, as well as more extreme downpours with a higher risk of flooding' (Defra 2008). Flood protection and biodiversity will be lost as wetland habitats are destroyed or dry out (Defra 2008). Increased transpiration and evaporation and reduced rainfall in some regions will put further pressure on remaining wetland habitats, resulting in lower soil water table levels in late summer/autumn (pers. comm. Mike Acreman 2007, in Hume 2008). Increased frequency of winter/spring flooding or summer droughts could also have a detrimental affect on the suitability of remaining grazing marsh habitats (Ausden *et al.* 2001, Milsom *et al.* 2002).

In addition to the changes in climate, over 50% of grade one agricultural land, predominantly in the southeast of England, will be at risk of flooding due to rising sea levels (NFU 2005). Flooding, particularly by saline water, and waterlogging have major implications for land use, farming practices, productivity and farm incomes to the point where farming futures will be threatened (Morris *et al.* 2003). Although most grasslands are in the west, rising sea levels will have a significant impact on coastal grasslands and grazing marshes (e.g. the extensive grazing marshes in Kent and Sussex).

OPTIONS FOR GRASSLAND MANAGEMENT

We consider here potential land-management options for grassland and how they may, or have been shown to, affect wading bird species. We assess how these relate both to land managed specifically for conservation and land-management options which are more likely to be useful when applied to the wider countryside.

Protected areas

Raising water levels is known to affect waders positively. However, there are negative consequences for crop yields and so this type of option is more likely to be used in protected areas than in the wider countryside. There are several ways to raise water levels, such as ditches and rills.

Ditches, modified to stay wet longer through the use of water-retaining structures such as penning boards, sluice gates or bunds, could display many of the same attributes as ponds (Bradbury & Kirby 2006). The *Wetting Up Farmland for Birds and other Biodiversity* project is currently examining the use of bunded (dammed) ditches (Table 1) to provide wet features for farmland birds (Bailey *et al.* 2007). Bunded ditches retain water and create wet areas alongside fields, creating greater availability of damp soil and more areas of permanent water, thus making water more available at critical times during the year. Ditches may be beneficial to farming systems by providing water for irrigation and stock.

Arterial drainage infrastructure influences the ability to manage water tables in field centres (Armstrong & Rose 1999). Eglington *et al.* (2008) described how by using a system of pumps and sluices, water levels could be raised in ditches and fed out into the centre of the grazing marshes using footdrains. Water levels can be raised to over-top footdrains, creating a mosaic of unflooded grassland interspersed with wet features and areas of shallow surface water, favoured by waders during the breeding season. An important feature of footdrains is that they provide a high level of control over surface water and cause little disruption to activities such as livestock management and sward production (Eglington *et al.* 2008), offering a management option that could be used on lowland wet grassland sites in the wider countryside. Footdrains could also act as water storage during drought periods and drainage channels during flooding events (Eglington *et al.* 2008).

Wider countryside

There are a range of options at different scales which could be used in the wider countryside. Flood risk management often relies on multiple management options, co-ordinated within a large catchment. The placement of these management options offers the potential for gains for biodiversity, such as waders. We begin by looking at the small-scale options and then move on to the co-ordinated catchment scale options.

Small-scale solutions

Ponds serve two purposes: pollution control and reduction of flood risk. Their effectiveness in controlling pollution is subject to location. For example, upstream wetlands (Table 1) trap few nutrients, whereas downstream wetlands, in key watershed positions, can remove up to 80% of inflowing nitrates (Crumpton *et al.* 1993, cited by Zedler 2003). Yet, during large storm events, a number of small wetlands strategically placed in the upper reaches of catchments will have a greater cumulative nutrient interception rate and be more cost effective than larger downstream structures (Raisin *et al.* 1997). A combination of approaches may therefore be necessary.

The use of farm ponds is being increasingly encouraged to mitigate diffuse, land-based sources of pollution due to their ability to retain nutrients. Vegetation inside ditches has been shown to enhance mitigation of the impacts of herbicides and some insecticides (Moore *et al.* 2001). Sustainable Drainage Systems (SuDS), designed to manage run-off associated with urbanized areas (e.g. roads), regulate flow rate and water quality in stages. Techniques include the use of vegetated filter strips and swales (channels), retention ponds and wetlands, and have been shown to be effective in the filtration and sedimentation of pollutants (Lawrence *et al.* 1996). SuDS also make a significant contribution to macro-invertebrate biodiversity (Scher & Thiéry 2005, LeViol *et al.* 2009) and have the potential to provide habitat corridors and refuges. In agricultural landscapes, Small Constructed Wetlands (SCWs) (Table 1), pond-like structures designed to promote the filtration and sedimentation of run-off in a similar way to SuDS, have been found to be very effective at reducing nitrogen export in sub-surface drainage from cattle-grazed pasture (Tanner *et al.* 2005) and if the subsurface water originates from hill slopes, the nitrate content can be reduced by up to 97% (Haycock & Burt 1993). When a number of ponds are placed in sequence, those receiving water that has been previously filtered may be of higher ecological value than those higher in the catchment (Stoate 2003). Nutrient interception and habitat quality may decrease as sediment accumulates and excess vegetation develops. Performance will also vary seasonally (Thorén *et al.* 2004) and with changing hydraulic and pollutant loadings (Fink & Mitsch 2004).

Surface waters, such as streams, remain in ‘good ecological status’ (only slightly deviating from conditions expected in the absence or near absence of anthropogenic impacts) until agriculture exceeds 30–50% of the catchment area (Allan 2004). In Britain, permanent and temporary grasslands occupy approximately 7 million hectares, over 65% of the agricultural land (MAFF 1997). Vast areas would need to be de-intensified to reach the maximum threshold of 30–50% agricultural use of a catchment area. This is impractical where agricultural production is the primary goal. Davies *et al.* (2008a) contrasted catchment characteristics among different water body types and concluded that de-intensification of agriculture at the scale of pond ‘microcatchments’ is more feasible and effective than it is on the catchment scale of larger aquatic systems, such as rivers or lakes. To attain ‘good ecological status’, an average pond requires only 4 ha to be de-intensified, compared with 10 086 ha for a river (Davies *et al.* 2008a).

In a modelling exercise, Heathwaite *et al.* (2005) found that small ponds that store water temporarily at the bottom of a field were effective in reducing overland flow following storm events. SuDS have demonstrated this ability (Mance *et al.* 2002, White & Howe 2002, Scholz 2003) and in Belgium, retention ponds were found to be very effective, reducing the peak discharge and total runoff volume by 40% (Evrard *et al.* 2007). It is likely that SCWs will function in a similar way. Small wetlands located high up in the catchment are also effective (Potter 1994). However, the value of small, widely distributed wetlands for flood control is dependent on the amount of storage relative to the volume of floodwater, as well as their capacity for evapotranspiration and infiltration (Potter 1994).

Strategic placement of pond type structures therefore offers reduced flood risk, but what gains can they offer for waders? On lowland grassland, high densities of breeding waders are associated with wet features and ponds with large perimeters and shallow sloping edges that provide significant areas of bare, damp soil suitable for foraging and habitat for obligate aquatic invertebrates. It is possible, therefore, to create ponds in all parts of Britain, including intensively managed agricultural landscapes. Williams *et al.* (2008) suggest that it may be possible to influence national breeding populations of wading birds through the development of a number of small-scale pond creation schemes within grassland systems. In addition, preliminary results have shown that established paired ponds (Table 1) are likely to be an important habitat for a wide range of non-wading bird species, as they retain water for longer than conventional unbunded ditches (Bailey *et al.* 2007). The schemes would consist of a waterbody mosaic in floodplain grassland including ponds that differ in size, substrate, water source and hydrological regime. Pond creation is most likely to be of benefit in open areas, such as field centres, where some wetland habitat already exists, for example alongside rivers, in existing areas of damp grassland, or beside reservoirs and gravel pits. In these areas, some feeding habitat may already be available even when habitats are unsuitable for breeding.

Large-scale solutions

Wetlands Floods can be controlled or prevented through the 'complementary roles' played by wetlands of varying sizes and at different locations (Zedler 2003). For example, large wetlands located low down in the watershed can be managed to reduce peak flood levels (Potter 1994). In addition to flood mitigation, wetlands also regulate river flows and promote groundwater recharge, although the capacity to perform these functions varies across wetland types (MEA 2005).

The surface area, depth and shoreline complexity of new wetlands can also be constructed to aid both nutrient retention and biodiversity (waders and their prey). Shallow, large wetlands with high shoreline complexity are likely to attract waders and have high macro-invertebrate biodiversity (Thiere *et al.* 2009) and nitrogen retention (Hansson *et al.* 2005). Conversely, small deep wetlands are less valuable for biodiversity but will have more efficient phosphorus retention (Hansson *et al.* 2005).Therefore, dual-purpose wetlands with high nutrient retention may not have a high potential for increasing biodiversity and vice versa (Zedler 2003, Hansson *et al.* 2005).

A recent study concludes that, on average, ecosystems take approximately 50 years to recover from agriculture (Jones & Schmitz 2009), and that the stochasticity of natural systems means that they may never return to levels found in pre-perturbation conditions. New wetlands are unlikely to perform the same functions or support the same biodiversity as historic wetland habitat (MEA 2005) as it is difficult to recreate conditions in areas where cultivation has altered topography, soil quality and biodiversity (Zedler 2003). To be successful, restored habitat must be sustainable, have comparable composition, productivity and nutrient retention to the 'target habitat' (Acreman *et al.* 2007), and be near to remnants of original habitats (Cedfeldt *et al.* 2000).

Washlands Washlands (Table 1) are typically found in areas of floodplain surrounded by river banks that provide a low level of flood protection (Morris *et al.* 2004). In a flood event higher than the banks, the washland fills with water and acts as a flood storage area, significantly reducing flood peaks downstream (Acreman *et al.* 2003).

By storing floodwaters in their soils or on the surface, washlands have the potential to provide wetland habitat, determined by the dominant land use on the washland and the catchment as a whole (Morris *et al.* 2002, 2004). This will be greatest in grassland or woodland areas that typically experience more frequent flooding and wetter ground conditions compared with arable land that requires infrequent flooding and drained soils (Morris *et al.* 2004). Wetness regime, substrate type, vegetation structure, grassland management and disturbance can influence washland habitat biodiversity (Joyce & Wade 1998) and variations in these factors can result in a mosaic of habitats (Morris *et al.* 2004).

Morris *et al.* (2004) describe three categories of washland: flood management washlands, integrated washlands and conservation washlands. These categories represent a range of flood management and biodiversity options. Where flood management is the primary objective of washland creation, biodiversity objectives will be met as long as they do not significantly compromise flood management purposes and vice versa. Integrated washlands give equal consideration to both. For breeding waders, flood duration and flood seasonality determine the suitability of a washland creation scheme and uncontrolled flooding can have a detrimental effect on breeding populations. For example, the Ouse Washes, designed originally for flood management, are now being managed as an integrated washland scheme (Morris *et al.* 2004). On integrated washlands emphasis is placed on the retention of surface water and soil wetness beyond the flood event period to create suitable habitat for breeding waders. However, since the 1980s, an increased frequency of flooding at the site has compromised biodiversity benefits. In particular, a dramatic decline in breeding Snipe and Black-tailed Godwits *Limosa limosa* L. has been attributed to an increase in the frequency of spring and summer flooding, effectively rendering the site unavailable during some breeding seasons (Ausden & Hirons 2002, Ratcliffe *et al.* 2005).

Conservation washlands may offer the best option for breeding waders. On conservation washlands, the creation of wetland habitats is the key objective and the frequencies, depths and timings of flood events are managed so as to maintain habitat quality (Morris *et al.* 2004). Prohibitive flood management regimes give rise to wetlands that function as reserves rather than truly multifunctional landscapes. As a result, individual wetlands may offer limited contribution to flood management. However, the cumulative effect of a number of wetlands over a whole catchment may be significant.

DISCUSSION AND CONCLUSIONS

Future grassland management can potentially, at a range of spatial scales, provide some solutions for both ecosystem services (water quality and flood alleviation) and grassland bird conservation. Factors influencing the use of wet features as foraging and nesting habitat by ground-probing birds are summarized in Table 2. At present, the Environmental Stewardship scheme (Natural England 2010) provides opportunities for the restoration, creation and maintenance of wet grassland for breeding waders. Wet feature creation is not currently included as an option. However, ponds offer good potential for both ecosystem services (through pollution control and reduced flood risk) and, if designed with gently sloping sides and placed in suitable areas, benefits to breeding waders. Higher Level Stewardship includes the option 'to provide additional flood water storage and flood defence through the restoration and recreation of wetland habitat for other objectives' (Natural England 2008). It also offers some possibility for the inclusion of catchment de-intensification as a method of improving the ecological condition of water bodies (Davies *et al.* 2008b). However, agri-environment schemes are taken up on a voluntary basis. Where areas identified for de-intensification or flood mitigation cross farm boundaries, cooperation between land owners and a co-ordinated approach would be necessary for success. A landscape-scale approach is essential to avoid creating isolated fragments of high-quality habitats (Benton *et al.* 2003, Whittingham 2007) and, particularly as climates change, landscapes will be required to be increasingly permeable to allow species to shift and adapt their ranges. A landscape-scale approach to placement of ponds is also crucial to maximize the benefit of flood risk and pollution control. It seems feasible that future new management options for farmland could include targeted schemes for both 'water' issues and biodiversity. These schemes may focus on different scales to the current Agri-Environment Schemes (AESs). For example, schemes at a local scale may be useful in protected areas (e.g. footdrains) and perhaps there is the opportunity to develop these within future high-intensity AESs. However, if future AESs are to be linked to wider ecosystem service goals and biodiversity in the wider countryside (i.e. outside protected areas) then they will need to address the issue of co-ordinated implementation at the appropriate scale. For example, the placement of ponds within a catchment needs careful planning to maximize both reduction in pollution, flood control and benefit for waders and this is not likely to happen if determined solely by land-owner uptake. Thought is needed as to how these types of schemes could operate and the input of social scientists may be needed to help with this issue.

We have focused this review on waders due to the relative lack of studies on other bird species identified by our literature survey. However, there is some evidence of the effects of soil moisture on other species. Between the mid-1970s and the early 1990s the UK Song Thrush *Turdus philomelos* L. experienced a significant population decline, with approximately 70% of pairs lost on farmland

Table 2. Factors influencing the use of wet features as foraging and nesting habitat by ground-probing birds.

	Factors influencing habitat use
Ponds	Retention of water and moist soil for probing during the spring and summer[a] Proximity to other wet habitats providing foraging and nesting habitat opportunities E, V, P, W, L
Ditches	Maintenance of water levels at mean field height throughout spring and summer to provide moist soil for foraging Hydraulic conductivity of the soil[b] E, V, P, W, L
Footdrains	Water levels in ditches feeding into footdrains E, V, P, W, L
Wetlands	Depth, size and shoreline complexity[c] Previous land use; e.g. to recreate productive habitat it is beneficial to choose restoration sites next to remnants of original habitats[d]
Floodplain washlands	Abundance of soil invertebrate prey in relation to frequency, seasonality and prolonged surface flooding[e,f] Spatial scale of flooding and availability of refuge for soil invertebrate prey[e,f] Previous field use; e.g. the flooding of previously fertilized grassland can result in anoxic conditions for soil invertebrates[e] V, P, W
Integrated washlands	The level of flood control during the nesting season; e.g. increased flooding frequency at the Ouse Washes has shortened the nesting season for Snipe[g] Abundance of soil invertebrate prey, as mentioned above[e,f] V, P
Conservation washlands	Wetness regime and vegetation structure suitable for foraging and nesting Grassland management and freedom of disturbance[h] V, P

E, sloping edges for foraging; V, vegetation swards for nesting; P, pollutant loading; W, frequency and seasonality of high water flow during the nesting season; L, perimeter length.
References: [a]Bradbury & Kirby 2006, [b]Gavin 2003, [c]Hansson *et al.* 2005, [d]Cedfeldt *et al.* 2000, [e]Ausden *et al.* 2001, [f]Plum 2005, [g]Ausden & Hirons 2002, [h]Joyce & Wade 1998.

alone (Baillie *et al.* 2001). Now only a fraction of the population lives on grassland. The timing and spatial distribution of the population decline is consistent with the pattern of land drainage in Britain, with the worst affected areas being the arable-dominated counties of eastern England (Peach *et al.* 2004b). Soil moisture is likely to be an important factor, but much of the effect may be in non-grassland areas. Although this does not necessarily infer causation of population decline it is consistent with this explanation. Compared with arable farmland, mixed farmland has areas of permanent pasture that retain damper soil conditions later in the breeding season, which in turn increases the length of time that earthworms, an important component of breeding season diet, are available (Gruar *et al.* 2003). During dry periods, provisioning adults forage further from their nest (Peach *et al.* 2004a) and the summer weights of chicks and adults are negatively related to the dryness of surface soils (Gruar *et al.* 2003). The duration of summer droughts is also negatively correlated with annual variation in adult survival rates, a key demographic rate (Robinson *et al.* 2004). Hot, dry weather is likely to affect Song Thrushes through the drying out of ditches and under hedges, thus reducing both above- and below-ground access and abundance of prey. Significant temporal and small-scale spatial variation in Chough *Pyrrhocorax pyrrhocorax* L. pre-breeding survival can also be linked to the effects of drier weather conditions on invertebrate prey abundance and accessibility (Reid *et al.* 2008). A range of other species, including the Mistle Thrush *Turdus viscivorus* L., also probe the ground for food, and soil moisture may act in similar ways for these species, although it is unlikely that the types of large-scale wet feature creation discussed here would be a viable option for wide-ranging and open grassland species.

Land use in the 'wider countryside' needs to integrate crop yield, ecosystem services and biodiversity if it is to be truly multi-functional (Firbank 2005). To date, the extent of research in this area is limited (e.g. Vickery *et al.* 1994). However, the combined pressure of global food production and climate change makes it questionable whether AESs in their present form can be sustained at high levels (Ausden & Fuller 2009). The schemes of the future may benefit from an integrated ecosystem services approach. By linking biodiversity objectives with other ecological objectives set out by policies such as the Water Framework Directive, conservation targets can continue to be met. At present, there is a lack of direct evidence of the quantitative impacts of management solutions on biodiversity and ecosystem services. However, there is the potential to help mitigate damaging effects of climate change and pollution and provide high-quality habitat for birds in both protected areas and the wider countryside via options such as appropriately designed and located ponds and for the latter (to a lesser extent) the use of integrated washlands. Not all water management options will benefit waders and the aims for any area need to be prioritized and co-ordinated at local, regional and national levels to maximize benefit for the different dimensions of land use.

One note of caution for future schemes linking biodiversity and ecosystem services is that ultimately the latter can be viewed in terms of 'benefits', such as clean drinking water, which are assessed in economic terms (Fisher *et al*. 2008). This carries with it some issues that impact on biodiversity. For example, the natural capital of the 'biodiversity' component may be identical but other factors may intervene. If a dam is built upstream in a water catchment, the measures (such as biodiversity) to alleviate flood control downstream are then of less economic value. The spatial location of the resource is important; for example, a wetland next to a source of pollution that can act as a filter is of greater value than one that is not (Vira & Adams 2009). Whilst the ecosystem services agenda is likely to impact on current AES policy the devil may be in the detail in terms of the benefit for biodiversity.

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