Rothamsted Repository Download

A - Papers appearing in refereed journals

Blackwell, M. S. A. and Pilgrim, E. S. 2011. Ecosystem services delivered by small-scale wetlands. *Hydrological Sciences Journal.* 56 (8), pp. 1467-1484.

Hydrological Sciences Journal

ISSN: 0262-6667 (Print) 2150-3435 (Online) Journal homepage: https://www.tandfonline.com/loi/thsj20

Ecosystem services delivered by small-scale wetlands

Martin S. A. Blackwell & Emma S. Pilgrim

To cite this article: Martin S. A. Blackwell & Emma S. Pilgrim (2011) Ecosystem services delivered by small-scale wetlands, Hydrological Sciences Journal, 56:8, 1467-1484, DOI: 10.1080/02626667.2011.630317

To link to this article: https://doi.org/10.1080/02626667.2011.630317

Ecosystem services delivered by small-scale wetlands

Martin S. A. Blackwell and Emma S. Pilgrim

Rothamsted Research, Department of Sustainable Soils and Grassland Systems, North Wyke, Okehampton, Devon EX20 2SB, UK
emma.pilgrim@bbsrc.ac.uk

Received 1 September 2010; accepted 25 July 2011; open for discussion until 1 June 2012

Editor Z.W. Kundzewicz; **Guest editor** M.C. Acreman

Citation Blackwell, M.S.A. and Pilgrim, E.S., 2011. Ecosystem services delivered by small-scale wetlands. *Hydrological Sciences Journal*, 56 (8), 1467–1484.

Abstract The benefits of small-scale wetlands have been largely overlooked, primarily because (a) such areas are considered problematic to manage, and (b) small wetlands fall outside the remit of most wetland inventories. The subsequent paucity of information prevents a comprehensive investigation of their properties and this must be addressed. Here we examine the evidence for the potential significance of small wetlands with regard to delivery of ecosystem services (ESs) and conclude that small wetlands often have a positive effect on their delivery, especially water quality, water regulation and biodiversity conservation. However these benefits can be offset by the emission of greenhouse gases. We suggest that, in future, wetlands should not be assessed on size alone, but rather in the context of both their location in the landscape and interaction with hydrological pathways. Furthermore, tools need to be developed to assess the type and efficiency of ESs delivered from all wetlands.

Key words wetlands; ecosystem services; biodiversity; water quality; water regulation; greenhouse gases

Services écosystémiques fournis par les petits milieux humides

Résumé Les avantages des petits milieux humides ont été largement négligés, surtout en raison du fait que (a) de tels domaines sont considérés comme problématiques à gérer, et que (b) les petits milieux humides se trouvent hors de la compétence de la plupart des inventaires de zones humides. De ce fait, la rareté des informations empêche une enquête approfondie de leurs propriétés, et cela doit faire l'objet d'études. Nous examinons ici la preuve de l'importance potentielle des petites zones humides à l'égard de services écosystémiques et concluons que les petits milieux humides ont souvent un effet positif sur les variables associées, la qualité de l'eau en particulier, la régularisation de l'eau et la conservation de la biodiversité. Ces avantages peuvent toutefois être occultés par l'émission de gaz à effet de serre. A l'avenir, nous suggérons que les zones humides ne soient pas évaluées sur leur seule taille, mais plutôt dans le contexte de leur emplacement dans le paysage et de leurs interactions avec les écoulements hydrologiques. De plus des outils doivent être élaborés pour évaluer le type et l'efficacité des services écosystémiques délivrés par toutes les zones humides.

Mots clefs zones humides; services écosystémiques; biodiversité; qualité de l'eau; régulation de l'eau; gaz à effet de serre

INTRODUCTION

Wetlands are some of the most productive and diverse ecosystems, both hydrologically and ecologically. Consequently, they are able to deliver a wide range of ecosystem services (ESs) of value to people. However, much of the research into wetland ESs has focused on large wetlands receiving protection under various designations. In contrast, small "patches" of wetlands that are often overlooked and unprotected, due to their omission from wetland inventories, can play a pivotal role in the delivery of a number of important ESs. These include: water quality regulation; hazard control (e.g. flood risk); numerous resources for human uses; habitats for plants, animals and micro-organisms; recreational opportunities; and the aesthetic value of the countryside. Much of this ability to deliver ESs arises out of their position within the landscape, as they are often located at significant positions along hydrological pathways where they are able to interact with waters draining agricultural land (Baker *et al.* 2009), or provide wildlife refuges within agricultural systems (Trochlell and

ISSN 0262-6667 print/ISSN 2150-3435 online

http://dx.doi.org/10.1080/02626667.2011.630317
http://www.tandfonline.com

Bernthal 1998). However, these positive attributes are frequently overlooked, because small wetlands have often been viewed as problematic in terms of agricultural production and, consequently, have been subject to land drainage (Acreman and McCartney 2009). There is also some evidence that suggests these wetlands may be significant contributors to greenhouse gas emissions (Hefting *et al.* 2006, Matthews *et al.* 2009, 2010) and, in some cases, increase the potential for flooding (Bullock and Acreman 2003). Here, we examine the evidence for the potential significance of small wetlands that are often overlooked or not considered with regard to delivery of ESs. Furthermore, we discuss the implications of the ensuing trade-offs between the delivery of more than one ecosystem service from small wetlands.

We consider small wetlands, typically less than 1 ha in area, including those that occur in the corners of agricultural fields (Fig. 1(a)), poached land that becomes saturated (Fig. 1(b)), ditches, wet hollows along hydrological pathways (Fig. 1(c) and (d)), and wet patches associated with discharge zones from either farmyard runoff, solid or liquid animal waste stores, or natural seepage zones (Fig. 1(b)). Ponds are not specifically dealt with here, although the distinction between ponds and wetlands is a grey area. While they are sometimes alluded to in this document, there are good reviews reporting the performance and management of ponds (e.g. Gustafson *et al*. 2000, Shilton 2008), and so the focus here is on the other types of small wetlands described above.

Defining wetlands

The definition and classification of wetlands is a complex subject and varies from country to country, reflecting their wide range of types, size and distribution. Typically, wetlands occupy the transitional zone between aquatic and terrestrial environments, have characteristically high water tables and experience periodic or long-term flooding. There are many definitions of wetlands, but generally they are accepted as having one or more of the following attributes:

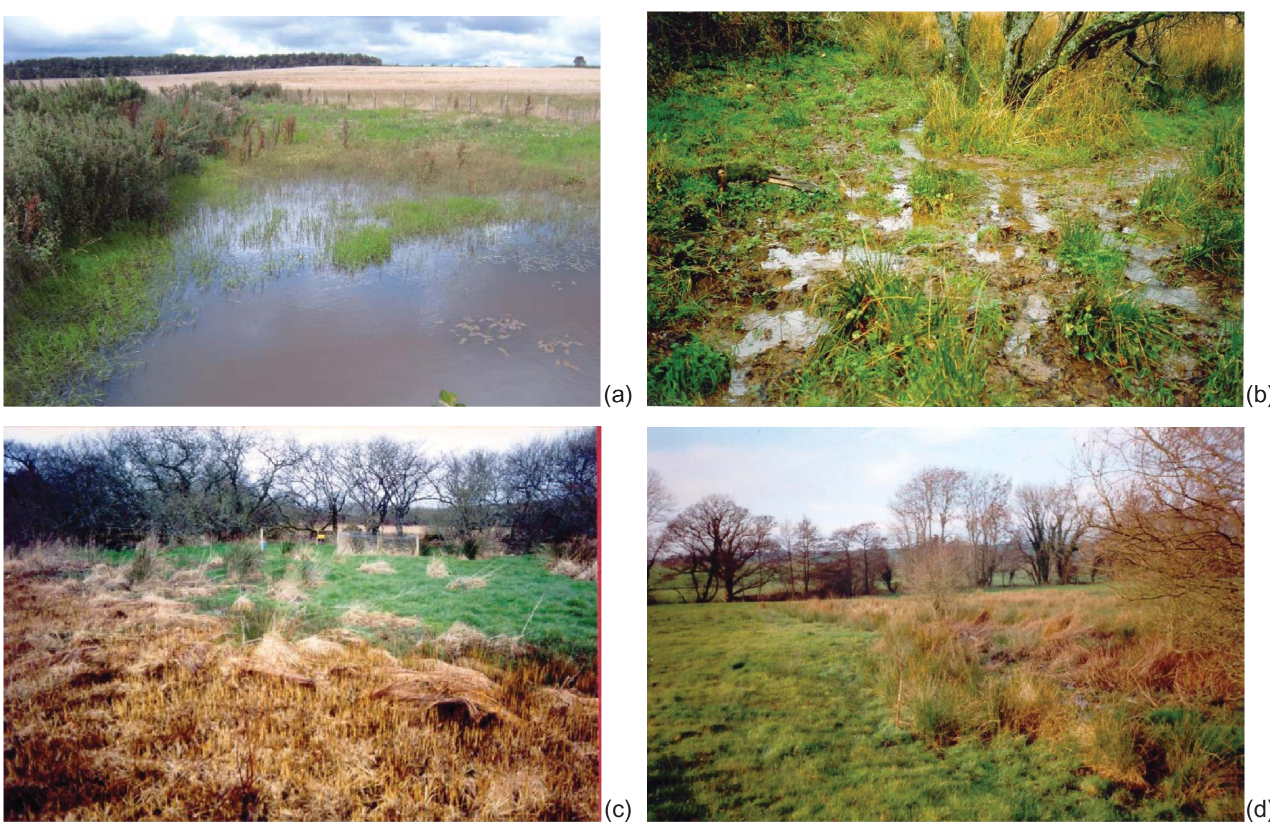

Fig. 1 Photographs showing: (a) an example of a wetland patch in the corner of a field; (b) an area of poached wetland seepage zone under woodland and scrub; (c) an area of overland flow (indicated by the bright green patch of grass) along a disrupted ditch network at Kismeldon Meadows, Devon, UK; and (d) a linear wetland formed in a former river channel across a floodplain.

– the presence of water either at the surface or in the root zone;
– unique (hydric) soil conditions different to adjacent, non-wetland areas; and
– vegetation adapted to permanently or seasonally wet conditions (Mitsch and Gosselink 2007).

The only international wetland designation system in use is that developed by the Ramsar Convention on Wetlands of International Importance Especially as Waterfowl Habitat (Matthews 1993, Scott and Jones 1995). This convention requires that contracting parties compile inventories of wetlands to enable the development of "wise-use" wetland policies. In the UK there are currently a total of 168 Ramsar sites occupying over one million hectares. Though sizes vary, typically these wetland areas are large; they include the Humber Estuary (38 000 ha), the Lewis Peatlands (59 000 ha), the Somerset Levels and Moors (6400 ha), as well as Turmennan Lough (15 ha) and Llyn Idwal (14 ha). This inventory system is designed to account for wetlands with regard to the single ecosystem service of habitat provision for waterfowl and, hence, smaller wetlands are not included.

Given the problems associated with defining and classifying wetlands, it is no wonder there are also problems with the development of wetland inventories. Comparison of wetland types in one country to those in another is often difficult because different classification systems are used, and many countries have national or regional wetland terminology that is not understood internationally (Scott and Jones 1995). Furthermore, the comprehensiveness of these inventories varies greatly. Despite this, the global extent of wetlands has been estimated to be between 7.0–8.5 $\times$ 10^6 km^2, or approximately 6% of the land surface (Maltby and Turner 1983, Mitsch and Gosselink 2007). In the UK, estimates of wetland extent vary, but it is estimated that over 90% of the original wetland extent has been lost due to drainage (Hume 2008).

Taking the UK as an example, there is still a large number of "wetlands" which are not recorded by most inventories. The main reason is that most land-use assessment exercises rarely consider a scale smaller than a field. The Countryside Survey (CS) (Carey *et al.* 2008), an approximately decadal survey of the vegetation and soil diversity in the British countryside, most recently conducted in 2007, is based on a stratified random sample of 1-km squares from the intersections of a regular 15-km grid superimposed on the rural areas of Great Britain. The CS has a minimum mappable habitat area of 400 m^2, which is reduced further to 20 m in length for linear features. Consequently, the record of the frequency of occurrence or distribution of those features that occur at less than a field scale is very limited. Moreover, some small wetlands can have ephemeral qualities and so, even if they are mapped at a suitable scale, depending on when the inventories are compiled, they still may not be included. While larger wetlands provide the largest proportion of Sites of Special Scientific Interest (SSSI) of any habitat in the UK, the small wetlands we consider here are perhaps the most vulnerable to degradation, given this failure to properly evaluate the importance of the services they provide "for free." This pattern is reflected in many agricultural countries around the world. For example, until recently many states in the USA regulated only wetlands that exceeded 4 ha, meaning that small isolated wetlands were extremely vulnerable to degradation (Kaiser 1998, Semlitsch 2000). While this has largely been addressed by new regulations, the biological and wider environmental importance of many small wetlands continues to be neglected, probably because of their size and ephemeral hydrology (Russell *et al.* 2002). Also, one of the reasons why small-scale wetlands fail to be considered is because they are often included as part of other habitat definitions and, subsequently, they are not recognized as being functionally different from their associated broader habitats. New approaches and tools are required to better identify, assess and evaluate the ESs delivered by wetlands, including small wetlands. One such tool, the Functional Assessment of Wetlands (Maltby *et al.* 2009), which breaks down the landscape into constituent hydrogeomorphic units, which can comprise features a matter of a few square metres in size, is described below.

While some of these small-scale wetlands may occur as a result of natural conditions, e.g. the wet patches in the corners of fields (Fig. 1(a)), or former channels (Fig. 1(d)), many are the result of anthropogenic actions or livestock behaviour (Fig. 1(b) and (c)). Despite their "artificial" nature, they all possess the characteristics and, to some degree, the functional qualities of larger, naturally occurring wetlands, and therefore we argue that they should be considered in the same way. Indeed in many cases, the manner in which these "artificial" habitats are created results in them being located along key hydrological pathways, enabling them to optimally perform wetland functions that can be of great value for the delivery of many services. In many cases it is not their spatial extent that is important, but the degree of hydrological interaction that affects their ability to deliver services. As described by Blackwell *et al.* (2009), if the overall area of the wetland is small but the length of interface with agricultural land is long, processes such as denitrification are optimized (Haycock and Burt 1993). Subsequently, although small wetlands may not appear to be delivering significant ESs, we are now beginning to acknowledge their cumulative significance and importance as components of a larger system (Johnston 1994, Trochlell and Bernthal 1998).

Quantifying the occurrence and extent of small wetlands is difficult, due to the lack of detailed land-use inventories at a scale appropriate for their inclusion, as explained above. One example that gives an indication of the extent to which some of these small wetlands occur was produced in the Tamar 2000 Project (Hogan *et al.* 2000), in which, as far as possible, all wetlands of all sizes occurring within the catchment of the River Tamar, southwest UK, were mapped. Wetlands were classified according to their location on either a slope (SL) or floodplain (F), and further categorized depending upon soil type and hydrology, giving rise to eight types of wetland category. The properties and water-quality enhancing processes associated with these different wetland units are summarized in Table 1 (Blackwell *et al.* 2009). Summary statistics for all these wetlands occurring in the catchment are given in Table 2. Many of the wetlands mapped in this survey were less than 1 ha in area, although this survey does not necessarily include very small features such as minor seepage zones or ditches. Floodplain units dominate the type of wetlands occurring spatially, but SL1 units are by far the most commonly occurring (241 individual units), and many of these are likely to be less than 1 ha. No other database exists with which to directly compare these data, but only two wetlands occurring within this catchment are recognized as being important, and hence designated SSSIs.

Another example of the extent of occurrence of small wetlands is given by Semlitsch and Bodie (1998). They examined the occurrence of isolated depressional wetlands on the southeastern Atlantic coastal plain of the USA. Using a geographic information system data set, established remotely, they identified 371 individual wetlands ranging in size from 0.2 to 78.2 ha in a 782-km^2 area. More than 46% of the wetlands were less than 1.2 ha, although this is almost certainly an underestimation due to the fact that the lower level of detection of wetlands was 0.2 ha. Therefore, again many smaller wetlands will not appear on this inventory.

DEFINING WETLANDS AND THEIR ASSOCIATED ECOSYSTEM SERVICES

Defining ecosystem services

Ecosystem services, as defined by the Millennium Ecosystem Assessment (MEA 2003) are "*the benefits people obtain from ecosystems*." The MEA (2003) distinguished four classes of ES, described as provisioning (products of the ecosystem e.g. food), regulating, supporting and cultural services. Though these definitions are widely used, numerous studies have modified these definitions, due to the difficulty in distinguishing between the supporting and regulatory services, a fact acknowledged by the MEA authors (2003). Consequently, in a previous literature review on ESs (Pilgrim *et al.* 2010), we used the MEA categories as the basis to identify ESs delivered from agricultural grasslands, four of which are described in Table 3, namely: agricultural production, water quality regulation, hydrological regulation and biodiversity conservation (Pilgrim *et al.* 2010). The focus here is on these four ESs, because there is very limited published literature on other ESs in the context of the small wetlands we are considering.

Principal ecosystem services delivered by wetlands

Wetlands play an important role in hydrological regulation (Bullock and Acreman 2003), but because of their ability to improve water quality through processes such as denitrification and sediment retention, they have been described as "kidneys of the landscape" (Mitsch and Gosselink 2007). However, in performing these processes, greenhouse gases can be produced that affect air quality and climate (Hefting *et al.* 2006). Furthermore, they can enhance biodiversity conservation (Pilgrim *et al.* 2010) by providing habitats for plants and animals (Hillbricht-Ilkowska 2008). In some contexts they may be part of productive systems, hence providing biological products. Overall, the ability to deliver the ESs listed in Table 3 is dependent upon their individual characteristics (e.g. size and shape), properties and settings. Often, small wetlands are able to perform these functions more efficiently (in terms of area) than large wetlands, e.g. water quality improvement (Blackwell *et al.* 2009).

Table 1 Descriptions of wetland functional units occurring in the River Tamar catchment, UK, based on geomorphology, hydrodynamics, soil type, vegetation and the key processes occurring within them that affect water quality (Blackwell *et al.* 2009).

Unit name	Geomorphic position	Other features	Hydrodynamics	Soils	Vegetation
F1	Floodplain: level and/or elevation	Occasional depressions, slacks	Occasional flooding; seasonal groundwater in subsoil	Permeable, brown alluvial soils	Mesophile dry grassland, scrub and woodland
F2	Floodplain: level/depression	Ditches, slacks	Flooding; high groundwater table	Alluvial gley soils	Rush pasture, willow scrub and woodland
F3	Floodplain: backland/depression	Ditches, oxbows, abandoned channels	Permanently high groundwater, seepage and surface flow inputs; flooding	Humic and alluvial gley soils	Tall herb fen, willow carr, flood pasture
SL1	Base of footslope (0–2°), including tributary valley floors lacking alluvium	Ditches, water tracks	High groundwater and some surface flow	Slowly permeable stagnogleys	Tall herb fen, willow carr, patches of flood grass along water tracks, some improved to rush pasture
SL2	Footslope (0–2°)	Peat mounds	Strong groundwater discharge fringing alluvium, on lower slopes and in valley heads	Peat	Tussocky *Molinia* and bog
SL3	Footslope (0–2°)	Peat mounds, floating vegetation mats	Strong groundwater discharge fringing alluvium, on tributary valley bottoms, lower slopes and in valley heads	Peat	Tussock sedge, willow, flood grass
SL4	Footslope (2–5°)	Ditches	High groundwater and rain-fed with soils of low permeability, occasional seepage	Slowly permeable stagnogleys and stagnohumic gleys	Humid grassland, fen meadow, wet heath (includes Culm grassland), willow, improved land with rush pasture
SL5	Backslope (>5°)	Irregular surface, slumping	Seepage and overland flow	Gleys of varied permeability	Rush pasture and willow scrub

Table 2 Summary statistics for occurrence and size of different wetland units occurring in the Tamar catchment.

Wetland unit	Total area (ha)	Proportion of whole catchment (%)	Number of individual units	Mean unit area (ha)	Maximum unit area (ha)	Minimum unit area (ha)
F1	2454.3	2.650	19	129.1	1737.2	2.53
F2	590.3	0.638	55	10.7	81.5	0.03
F3	25.6	0.028	11	2.3	11.0	0.26
SL1	847.5	0.915	241	3.5	30.6	0.03
SL2	50.0	0.054	18	2.8	22.3	0.28
SL3	1.8	0.002	3	0.6	0.9	0.25
SL4	567.4	0.613	74	7.7	53.2	0.40
SL5	32.9	0.035	10	3.3	6.4	0.65

Agricultural production

Many of the small wetlands considered here occur within agricultural settings, even within productive fields themselves, and, therefore, deliver the ecosystem service of agricultural production depending on what crop is being grown. However, in more extensive agricultural systems they can be of true agricultural benefit. In Alberta, Canada, Sankowski *et al.* (1987) report that, in a complex of 65 small wetlands, species such as *Eleocharis palustris* (spikerush) provide over three times as much forage than the unmanaged grasslands around them, and that it also contains 77% higher protein concentration. However, there can be detrimental impacts resulting from small wetlands with regard to productivity. Wetlands can be hosts to numerous insects and invertebrates, some of which are vectors of disease or parasites. In East Anglia, UK, the development of small wetlands and wetter pastures associated with the Environmentally Sensitive Area scheme was associated with increases in liver fluke (*Fasciola hepatica*) infection in cattle, resulting in weight loss and decreased milk yields (Pritchard *et al*. 2005). As grazing of wetlands for biodiversity and aesthetic reasons increases across Northern Europe, increasing problems of this sort are being found (Begg 1986, Thamsborg *et al.* 2010).

Agricultural production on more than 30–50% of a catchment is reported to result in river water quality degradation (Allan 2004). Consequently, vast areas of land would need to be taken out of agriculture in most European countries if this threshold were to be met. As this scenario is unfeasible, Davies *et al.* (2008) suggest that focusing such reductions of agricultural land on the catchments of small wetlands would require relatively small quantities of land to be taken out of production. However, because of the important and disproportionate contribution they make to regional aquatic biodiversity, it would mean that aquatic biodiversity as a whole would reap great benefits. They refer to this targeting of the catchments of small wetlands, as opposed to whole river catchments, as a "micro-catchment" approach, which enables "pockets" of high aquatic biodiversity to occur within working agricultural landscapes.

Hydrological regulation

Most wetlands are inherently part of a wider hydrological system; consequently, the presence of wetlands, their type and the way they are managed will almost certainly influence some part of the water cycle. Typically wetlands are situated in low-lying areas, often on significant hydrological pathways; therefore, they can affect storm and flood water dynamics. However, the way in which this influence manifests itself can be complex and variable (Bullock and Acreman 2003). Though the cumulative effect of many small wetlands can be significant at a (sub-) catchment scale (Potter 1994), this is dependent upon their evapotranspiration and infiltration rates. Evidence suggests that some small wetlands can have higher rates of evapotranspiration and, therefore, are more efficient than larger wetlands at reducing runoff (Knight 1993, Millar 1971). Heathwaite *et al.* (2005) found that small wetlands that temporarily store water in agricultural landscapes were effective in reducing overland flow following storm events, while Evrard *et al.* (2007) report decreases in peak discharge of 40% in small catchments in Belgium through the development of water retention ponds. However, the value of small wetlands for floodwater control is dependent upon their distribution and the quantity of storage relative to the volume of floodwater, as well as their capacity for infiltration and evapotranspiration (Potter, 1994). In Florida, USA, the potential water storage capacity of small isolated wetlands was estimated by Lane and D'Amico (2010) using LiDAR data. They calculated a mean

Table 3 Description and examples of ecosystem services (ES). Information adapted and reproduced from Pilgrim *et al*. (2010).

Service	Description	Examples
Agricultural production	Agricultural production is defined as the extraction of biological products and services from ecosystems that are innovated and managed by people (following McIntyre *et al.* 2009). This ES contains all of the provisioning services described in the MEA (2005a, 2005b). With regard to energy we focus on bio-energy production comparing bio-crops, e.g. *Miscanthus, Panicum virgatum* (switch-grass), with biogas derived from anaerobic digestion of plant material or livestock wastes.	Food (meat, milk), fibre (e.g. cotton), fuel (e.g. wood, biofuel).
Water quality	Historically, regulations have focused on chemical determinands, though now the quality of open waters is assessed by using both ecological and chemical methods (Dodkins *et al.* 2005). The transferral of pollutants from the land to the water typically follows a mobilization–transport–delivery continuum (Haygarth and Sharpley 2000), but it should be noted can occur from both diffuse (from the managed land) and point (from Sewage Treatment Works or farmyard hard-standings) sources.	In grassland-dominated systems water quality is affected by the loss of nutrients primarily, nitrogen (N) and phosphorus (P) in addition to carbon (C), sediment and pathogenic organisms from land-based activities to surface and ground waters.
Hydrological regulation	Changes in land management can influence the water cycle by changing: hydrological flow paths and rates, storage capacity in the soil, aquifer interactions as well as plant uptake, storage and release (MEA 2005a).	Storage of water during rainfall events can help alleviate downstream flooding.
Biodiversity conservation	Here we specifically address terrestrial and freshwater aquatic ecosystems and the ecological complexes of which they are part. Most of the research is at the whole organism or the assemblage level, though the diversity of genes, populations and species underlies all grassland ecosystem processes (MEA 2005a).	Wetlands such as calcareous fens provide habitats for specialized and rare plants and animals.

value of 1619 m^3 ha^{-1}, and claimed their results could be used for hydrological modelling at the landscape scale to estimate ecosystem services. As a result of the connectivity between small wetlands and the wider hydrological system, Grenfell *et al.* (2005) demonstrated that small wetlands can be used as indicators of hydrological change within catchments. They observed a decrease in the extent of wetland vegetation in a small hillslope seepage wetland in South Africa, following the conversion of much of the upslope area from grassland to commercial forestry. In a review of the impacts of agricultural drainage on aquatic ecosystems, Blann *et al*. (2009) found that the drainage and connection of formerly isolated, small wetlands, causing the development of linear systems rather than wetland mosaics, resulted in damaging increases in flows in downstream rivers.

Water quality

Some small wetlands with standing water or dense emergent vegetation have the ability to slow the passage of water, and trap pollutants associated with sediments, such as phosphorus (P) and heavy metals (Kadlec and Knight 1996, Blackwell and Maltby 2006). This ability to trap sediment means that nutrients and other pollutants can accumulate in wetlands, giving them the potential to be highly productive ecosystems. Consequently, plant uptake of potentially polluting nutrients can play an important role in their ability to enhance water quality, particularly during the growing season (Picard *et al.* 2005). However, in winter when plant growth is minimal, these nutrients may be released into the environment in less reactive organic forms (Van der Valk *et al.* 1979). Furthermore, loading wetlands with nutrients can result in loss of biodiversity and dominance of invasive species (Rutchey *et al.* 2008).

The physical, chemical and biological function of wetlands can all contribute to enhancing water quality. For chemical processes, such as denitrification, the length of the interface and the degree of interaction with polluted surface waters and retention times (which are largely controlled by the position of the wetland in the landscape) are both more important than wetland size (Knight 1993). Haycock and Burt (1993) demonstrated that only relatively small areas are required for removal of nitrate (NO_3) in agricultural runoff, as complete denitrification occurred within 5 m of the interface between agricultural land and a downslope wetland.

In New Zealand, small wetland hollows with organic rich soils occurring on hydrological pathways that drain agricultural land were observed to remove 56–100% of the NO_3 in the drainage water (Cooper 1990), and denitrification rates at their upslope edge were two orders of magnitude greater than downslope rates. This was thought to be due mainly to lower NO_3 concentrations away from the upslope interface, limiting rates of denitrification. Small riparian wetlands with springs have also been shown to be significant for the removal of NO_3 from agricultural runoff, despite it commonly being thought that, due to the upwelling of groundwater, interaction times would be too short between polluted runoff and the soil. Vertical mixing means that much more interaction occurs than previously considered, meaning relatively high rates of denitrification can occur (Rutherford and Nguyen 2004). However, at high NO_3 concentrations, high nitrous oxide (N_2O, a powerful greenhouse gas) emissions were observed (Zaman *et al.* 2008), demonstrating the problems these types of systems can present with regard to swapping pollution from one form to another (see below). Other small wetland features, such as disrupted ditches, or even managed ditches themselves, can provide considerable water-quality benefits. For example, Blackwell *et al.* (1999) describe a discrete ditch system within a larger wetland at Kismeldon Meadows in the River Torridge catchment, southwest England (Fig. 2) that provides an example of how a small wetland feature can perform the service of water purification. A ditch draining improved agricultural land and that also passed through the wetland had become disrupted at several locations along its course, forming discrete areas of overland flow. Monitoring of nitrate (NO_3^-) concentrations in the ditches indicated that more than 90% of the NO_3^- in the ditch water is removed regularly in these zones, with the lowest observed removal efficiency being 60% (Blackwell *et al.* 1999). These small wetland areas are therefore performing the service of water purification by the removal of NO_3^- by a combination of both plant uptake and denitrification. Evidence exists that small wetlands can also improve the quality of water that contains pollutants such as surfactants (e.g. linear alkylbenzene sulfonates, LAS). In a wetland measuring only 474 m^2, Inaba (1992) reported that approximately two-thirds of the LAS flowing into it could be removed annually. Vegetated ditches have been reported to enhance the mitigation of the impacts of herbicides and pesticides (Moore *et al.* 2001). While, in some cases, individual small wetlands can provide substantial services

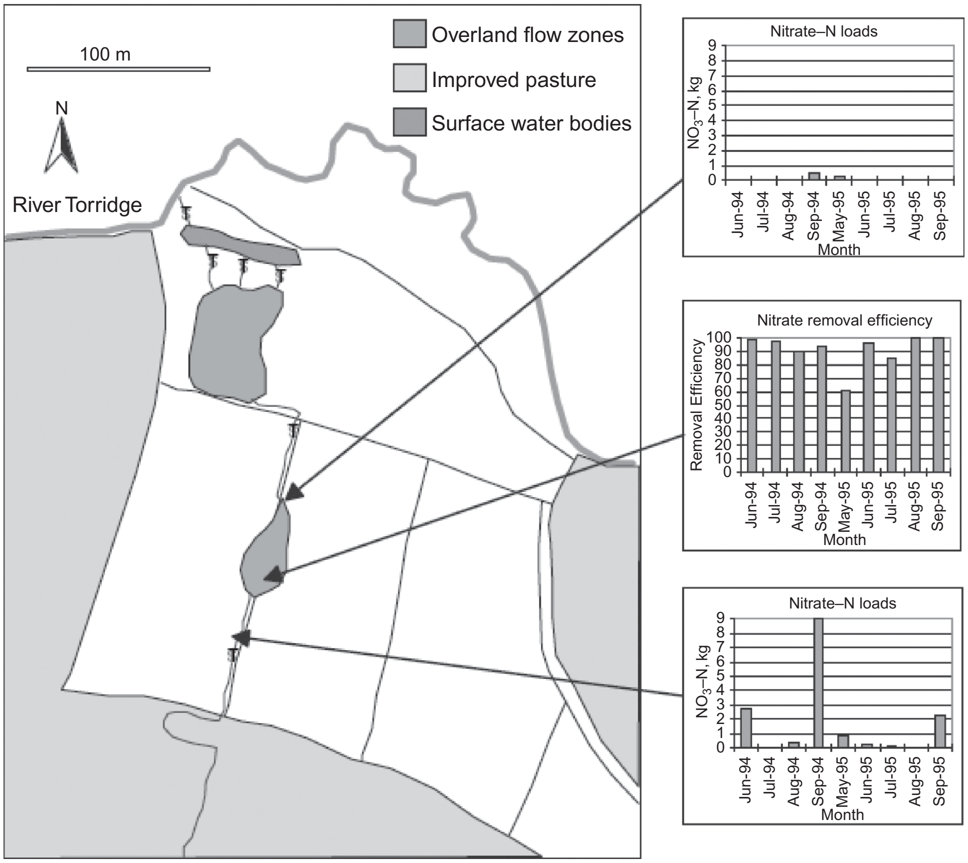

Fig. 2 Map of ditch network and nitrate concentrations at Kismeldon Meadows (after Blackwell *et al*. 2009).

and, in particular, water quality benefits, often it is the cumulative effect of several, linked small wetlands that provides the full benefit. This fact is recognized in Australia, where sheep farmers are actively encouraged to maintain small, linked in-stream wetlands along stream systems (Lovett and Price 2006). Among the benefits recognized as being delivered by these systems are water quality improvement, habitat provision and water quality regulation. It is estimated that they trap and retain approximately one third of the eroded sediment entering the streams, equivalent to 160 t per linear kilometre (Zierholz *et al.* 2001).

Further examples of small wetlands that can deliver the service of water purification are given by Blackwell *et al*. (2003). This study included an economic valuation of the service of water purification in relation to NO_3^- removal by measuring denitrification rates in several small wetland areas on a dairy and sheep farm. Relatively high actual and potential reductions in NO_3^- concentrations in soil water through the wetlands were measured, coupled with high denitrification rates. The values of the wetlands for denitrification alone were estimated at approx. £58 ha^{-1} $year^{-1}$, compared with £87 ha^{-1} $year^{-1}$ for sheep production. While lower than the commercial value for sheep production at the time, if a full economic valuation of the additional water quality benefits were carried out, as well as of the wider environmental benefits delivered by the wetland, it is likely that this value would be considerably greater. However, we must be aware that, where the land is managed for the delivery of one ES, e.g. water quality, there may be unintended negative consequences for other ESs, e.g. climate regulation, where a by-product of denitrification is greenhouse gas (GHG) production (see below). Such trade-offs arise as ESs are not independent of each other (MEA 2005a, 2005b)

Biodiversity conservation

Wetlands incorporate unique habitats with endemic and migrant species, for example, breeding and migratory birds (Matthews 1993), amphibians (Wilbur 1984, Duellman and Trueb 1986, Wind and Beese 2008), plants (Hajek *et al.* 2006), bees (Moron *et al.* 2008), molluscs (Hajek *et al.* 2006), and, therefore, contribute to the conservation of high global diversity (Zacharias *et al.* 2003). Many small, structurally simple wetlands are important habitats for maintaining the diversity of invertebrate and amphibian populations that are vital components of larger food webs (Richter and Azous 1995). Those small wetlands which regularly dry down on a seasonal basis can fulfil the dual purpose for the amphibians of providing unviable habitats for predatory fish and invertebrates, ensuring successful amphibian reproduction (Wilbur 1984, Duellman and Trueb 1986), and enhancing primary productivity through fluctuating water levels that can also increase food production. Around the world, small ephemeral wetlands shelter many extremely rare and isolated taxa, and give rise to allopatric speciation (Deil 2005).

Russell *et al.* (2002) identified small isolated wetlands (0.38–1.06 ha) as being focal points of herpetofaunal richness and abundance in managed coastal plain forests in South Carolina, USA. They identified 20 amphibian and 36 reptile species in these wetlands, and concluded that they contribute more to regional biodiversity than their small size and ephemeral hydrology implies. Also in the USA, Moler and Franz (1987) suggested that toads living in and around a small 1-ha isolated wetland could support a snake population occupying over 1000 ha of upland habitat. This is just an indicator of the importance of such small isolated wetlands and their potential role in food-web dynamics. It also indicates how their loss could impede source–sink processes and increase the likelihood of population extinctions at remaining wetlands due to over-predation (Semlitsch and Bodie 1998, Russell *et al.* 2002).

As with water quality benefits, the total area of wetland features is not always the most important factor determining a wetland's value, but its shape can have important influences on biodiversity factors. Smart *et al*. (2006) showed that breeding Redshank (*Tringa totanus*) density was directly proportional to length of wet features, such as drains, pools and ditches, while Eglington *et al.* (2008) showed that Lapwing (*Vanellus vanellus*) density was strongly related to the occurrence of ditch-like features. The small wetlands described by Sankowski *et al.* (1987) not only provided improved forage for livestock (see above), but also provided habitat for nearly 2000 breeding pairs of ducks, in contrast to reported declining population numbers reported elsewhere at the time. The importance of small wetlands, often unrecognized as important for conservation, has been identified in southern Africa as providing valuable feeding sites and migration staging posts for both lesser and greater flamingos (McCulloch *et al.* 2003).

Small wetlands with connections to surface waters can provide important spawning and nursery grounds for a number of fish species. Features which could act in this capacity typically include disconnected river meanders on floodplains and semi-natural ditches connecting land drains to surface waters. Neither, however, would generally appear on a wetland inventory. There is increasing interest in the management of ditches for the delivery of wetland ecosystem services. Despite their small size, bunded ditches can retain water and effectively act as small, linear wetlands, with many benefits, especially for wildfowl (Bradbury and Kirby 2006).

Calcareous fens, which are defined as groundwater discharge wetlands (Hajek *et al.* 2006), are listed under the EU's Habitat Directive Annex 1 as a priority, protected habitat, as they can have extremely high biodiversity and rarity value (Wassen *et al.* 2005). These habitats, which are typically less than 0.25 ha in size, are fed by up-welling groundwater rich in calcium and magnesium leading to alkaline soils in which only a small group of unique and rare calcium-tolerant plants can survive (Wolfe *et al.* 2006). The pothole ecosystems occurring in the Masurian Lakeland in Poland are also ecologically important, small (typically <1 ha) isolated wetlands supporting over 20 different plant communities (Wilpiszewska and Kloss 2002).

Other ecosystem services performed by small wetlands

Wetlands, ponds, hedges and managed ditches are all landscape features that enhance farmland biodiversity e.g. (Haycock and Muscutt 1995, Burel 1996, Viaud *et al.* 2005), as well as water quality and hydrological regulation (Hillbricht-Ilkowska 2008). Moreover, a diverse array of habitats is aesthetically pleasing, as well as contributing to both water quality and hydrological regulation (Blackwell and Maltby 2006).

The key way in which small wetlands affect climate is via the production and emission to the atmosphere of greenhouse gases such as methane (CH_4) and nitrous oxide (N_2O). As discussed below in the Trade-offs section, the quantity of greenhouse gases produced by small wetlands can be disproportionately large (Matthews *et al.* 2009), and is affected by factors including length of interface with other systems and supply of nutrients (Haycock and Burt 1993, Blackwell *et al.* 2009). Wetlands can affect air quality by promoting the emission of gases such as ammonia (NH_3), but there is little evidence for natural small wetlands having any significant effect, and, generally, emissions are reported to be low (Wang *et al.* 2010). However, emissions from constructed treatment wetlands receiving high ammoniacal loadings can be high (Van der Zaag *et al.* 2008).

Trade-offs between wetland ecosystem services

Another gap in our knowledge occurs, as, generally, only singular functions of these small wetlands are reported, that focus on benefits, whereas, in reality, they will be performing a whole range of functions to greater or lesser degrees, some of which are beneficial (e.g. hydrological regulation), whilst others are detrimental (e.g. GHG emissions). This concept, where a mitigation measure introduced to reduce levels of one pollutant unintentionally results in an increased levels of another pollutant, is defined as pollution swapping, a phenomenon that is gaining more attention (Stevens and Quinton 2009). Here we discuss the benefits and trade-offs (or negative outcomes) delivered by these small, largely neglected wetlands.

The drainage of wetlands for agricultural production can have a negative effect on hydrological regulation (Fig. 3), leading to flooding downstream (Pilgrim *et al.* 2010), although this affect can be variable depending on the location and other characteristics of the wetland within a catchment (Bullock and Acreman 2003). Factors including soil properties, position in the landscape and rainfall patterns will also determine the impact of artificial drainage on agricultural land (Robinson 1990). Typically, water on drained land will move more quickly from the land to surface water bodies, enhancing flood risk (Pilgrim *et al.* 2010), though Bullock and Acreman (2003) argue that the flood-control function largely applies only to floodplain wetlands, whilst other wetland types might increase flood risk.

Furthermore, there is a clear inverse relationship between hydrological regulation and climate regulation (Beier *et al.* 2008), whereby reduction of hydrological regulation by draining land will lower GHG emissions, as there will be fewer wet areas in which denitrification can occur, so subsequently the production of nitrous oxide (N_2O), methane (CH_4) and carbon dioxide (CO_2) will be reduced (Groffman *et al.* 1991, Ambus 1998, Mander *et al.* 2005). Moreover, increasing global temperatures make it likely that ever-increasing quantities of CO_2 will be released from organic soils, which are frequently associated with wetlands, resulting in a positive feedback to global warming (Freeman *et al.* 2001, Clair *et al.* 2002). In contrast, maintaining flooding (for natural flood defence), on floodplains, will result in the development of backswamps which will subsequently promote GHG emissions (Bouman 1990). However, this is a complex relationship, as dry soils also emit nitrous oxides (NOx) and ozone (O_3) (Cardenas *et al.* 1993).

The use of wetlands to improve the quality of runoff from agriculture not only has implications for greenhouse gas emissions, but also can have negative effects on plant community biodiversity. According to Verhoeven *et al.* (2006), most wetlands can incorporate increases in nutrient loadings, but when loadings surpass a critical level, species composition can shift dramatically. Naturally nutrient-poor systems react more drastically than

Responding ES B →

Drivng ES A ↑	Agricultural production	Climate Regulation	Air quality	Water quality	Hydrological regulation	Erosion regulation	Nutrient cycling	Biodiversity conservation	Landscape quality
Water quality	↔	↕ **	↕ **		↕ **	0	0	↑ **	↑ **
Hydrological regulation	↕ **	↕ **	↔	↑ **		0	↑ ***	↑ **	↑ **

Fig. 3 Trade-offs between the deliveries of mulitple ecosystem services (ESs) from wetlands. The relationship between the ecosystem service A (ES A) driving the change and the response of the ecosystem service B (ES B) to this change. The meanings of the symbols are as follows—0: no direct relationship, ↓: decline in ES B, ↑: increase in ES B, ↕: evidence of the relationship between ES A and B is divided or inconclusive, ⟷: no current evidence in the literature of an interaction between ES A and B. The strength of the relationship between ecosystem A and B is reflected in the number of stars—***: highly confident about evidence, **: mixed confidence about evidence, *: poor confidence in evidence. Cell colour reflects scenario type—light grey: win–win, dark grey: lose–lose and bordered cell: variable outcome. For example, we are highly confident that increasing agricultural production in intensively managed grasslands causes a decline in air quality due to the production of, for example, ammonia (NH_3) and non-methane volatile organic compounds (NMVOC). This is a win–lose scenario, as we increase food production at the expense of air quality, which has human health implications. This diagram is modified from Pilgrim *et al.* (2010).

more eutrophic systems, and continued loading of nutrients also typically results in a breakdown of the nutrient retention function. There is, therefore, a fine balance that needs to be established between biodiversity conservation and water quality improvement when utilizing small wetland systems to treat agricultural runoff. An example of this is provided by the overland flow zone at Kismeldon Meadows, southwest England (see Figs 1(c) and 2), where high concentrations of NO_3 in runoff from agricultural land are reduced in this small wetland area, but the valued *Molinia caerulea* grassland has converted to a less valued *Glyceria fluitans* dominated sward, as illustrated by the green patch evident in Fig. 1(c) (Blackwell *et al.* 2009).

Case study of ES trade-offs in small wetlands

Matthews *et al.* (2009, 2010) provide another example of how small-scale wetlands can deliver ESs to a significant degree, but also with trade-offs. Focusing on potential areas for GHG production, they report that many features found within agricultural livestock farming systems, including small wetlands and features with wetland soil properties, are potentially conducive to high denitrification/N_2O emission. These include poached land surrounding feeding and water troughs, waterlogged (or wetland) areas, gateways, tracks, ditches, as well as seepage from liquid and solid manures. In and around these features, soil conditions are typically anaerobic due to waterlogging and compaction, have high NO_3^- availability from concentrated faecal and urine deposition, high organic matter content from faecal deposition and low decomposition rates, and low soil pH typical of many reduced, anaerobic environments. In combination, these properties suggest all these features have the potential to be denitrification and N_2O emission hotspots. While this can be of benefit, because it represents a mechanism by which potential pollutants can be removed from the hydrological system, as described above, it also represents a trade-off. This is because N_2O is both a potent GHG with a global warming potential approximately 300 times greater than CO_2 (Ramanathan 1998) and is implicated in the depletion of stratospheric O_3 (Crutzen 1970, Cicerone 1987). In 1990 Bouman estimated that N_2O has been responsible for approx. 5% of the total enhanced greenhouse effect over the past 100 years.

Matthews *et al.* (2010) report that higher fluxes of N_2O were emitted from small-scale farmland features compared with adjacent pasture land. Overall, the greatest annual N_2O emissions were recorded from poached land around feeding troughs, field-based solid manure heaps, as well as from areas of seepage from yards and liquid manure stores. This is attributed largely to coupled nitrification–denitrification (Reddy and Patrick 1984) where large quantities of N_2O and ammonium (NH_4^+) can be produced, despite there being little measurable NO_3^- in the soil (Reddy *et al.* 1990, Mitsch and Gosselink 2007).

Small wetland features often attract concentrated livestock activity for a number of reasons: waterlogged pasture may support preferential grazing material at certain times due to maintenance of plant growth during periods of drought, or, in the case of seepage zones, as a result of enhanced nutrient inputs. Such concentrated activity around these features leads to poaching and compaction of the soil, which enhances conditions conducive to denitrification. This is also associated with concentrated deposition of urine and faeces, which also can create small-scale hotspots of denitrification and N_2O activity (Yamulki *et al.* 2000, Petersen *et al.* 2004).

Matthews *et al.* (2010) highlight that small-scale farm wetlands and wetland-like features can be either seasonal or persistent hotspots of N_2O emissions, with both inter- and intra-feature high spatial and temporal variability. This variability also extends to the farm scale for a number of reasons, including differences in soil types, farm management and land use. In some situations, these hotspots contributed significant proportions of the total N_2O fluxes from farms. Currently, many of these sources are not considered by the Intergovernmental Panel on Climate Change methodology (IPCC 1997) for determining N_2O emissions from grassland systems. This means that, although the relative contribution of these features to whole farm emission is generally small, there could be significant underestimation of N_2O emissions in certain situations.

Similar results have been reported from the same farmland features for CH_4 production (Matthews *et al.* 2009). Currently, UK IPCC methodology (1997) considers only sources from enteric fermentation and manure management when calculating agricultural CH_4 emission. The emissions estimated for the four farms using this methodology, as well as those calculated for the small-scale features are shown in Table 4. Methane emissions from the small-scale features on Farms II and III were equivalent to 14.2 and 16.9% of the total estimated by the IPCC

Table 4 Estimated emissions of CH_4 from IPCC recognized sources on the farms studied by Matthews *et al*. (2009), using UK IPCC methodology revised 1996 guidelines (IPCC 1997).

t CH_4 per farm	Produced as by-product of enteric fermentation	Emissions from animal manures	Total farm emission (IPCC)	Total emission from small-scale features
Farm I	21.42	3.90	25.32	22.06
Farm II	16.52	3.40	19.98	2.83
Farm III	20.36	1.64	21.99	3.71
Farm IV	6.29	0.64	6.92	10.92

approach, respectively. However, on Farm I, primarily as a result of the large area of seepage from a yard and a liquid manure store, total emission from features was equivalent to 87.1% of the IPCC estimate. Furthermore, on Farm IV, emission from small-scale features was more than 1.5 times greater than the IPCC whole-farm emission estimate. These are substantial losses of CH_4 that are not accounted for within the current IPCC methodology, and this highlights the importance of including soil-based on-farm sources in emission estimates, particularly water-saturated areas where an accumulation of substrates may occur.

Small-scale wetlands and buffer zones

"Buffer zone" is a generic term referring to naturally or semi-naturally vegetated areas situated between agricultural land and a surface water body (Blackwell *et al.* 1999). The capacity for small wetlands to act as effective buffer zones is well documented, and illustrated by the increasing development of small constructed wetlands for the treatment of domestic and industrial wastewaters (Brix 1994, Cole 1998). The ability of wetland buffer zones to remove 75% or more of the NO_3 from agricultural runoff via denitrification in relatively small areas has been well documented (e.g. Peterjohn and Correll 1984, Cooper 1990, Blackwell *et al.* 1999, Hefting *et al.* 2003). In the UK, the role of buffer zones for surface water protection is now recognized within Environmental Stewardship with an option for riparian buffers of up to 24 m width. Many of these buffers will be wetlands, due to their location near surface water bodies and at the bottom of hills, although sometimes they may comprise dry soils, depending on the parent material and landscape position. Generalized hydrological and geomorphological relationships between riparian buffer zones and whether or not they are effectively "wetlands" are shown in Fig. 4.

In contrast to nitrogen (N), the evidence for phosphorus (P) removal in small wetlands and buffer zones is limited. As described by Dorioz *et al.* (2006), removal tends to be primarily through the retention of sediment to which P is bound. However, Dunne *et al.* (2007) suggest that the storage potential for P in small (approximately 1–2 ha), isolated wetlands in Florida could be increased by adding N to the system, as the wetland biomass was N-limited. This would promote the accumulation of soil organic matter, and effectively "lock-up" P in more benign, organic forms. However, several researchers report that, for dissolved P, buffer zones can sometimes be net emitters, with retention ranging from –80% to +95% (Uusi-Kamppa *et al.* 2000, Duchemin and Madjoub 2004, Stutter *et al.* 2009). The key processes associated with N removal in buffer zones usually involve some form of transformation and potential emission (e.g. via gaseous N emission following denitrification), meaning N removal is effectively sustainable. Concerns remain for the long-term effectiveness of P removal from small wetlands and wetland buffer zones, especially with regard to sediment-associated P.

There is also potential for buffer zones to increase GHG emissions (Stevens and Quinton 2009). For example, riparian buffer zones produce greater quantities of N_2O than field margins, whilst wetland forested buffers produce seven times more N_2O than grassed ones (Hefting *et al.* 2003).

Tools for assessment of wetland ecosystem services

Given the growing interest and acknowledgement of the multiple ESs that wetlands can deliver, and the potential trade-offs of these services, there is an ever increasing need to develop tools to assess the type and extent of ecosystems services deliverable from wetlands of all sizes. One such tool is the Functional Assessment of Wetlands developed by Maltby *et al.* (2009). This tool enables both experts and non-experts to assess the functions (which equate to ESs) a wetland is performing. It is a field- and desk-based exercise with the smallest-scale unit considered being a hydrogeomorphic unit, which involves

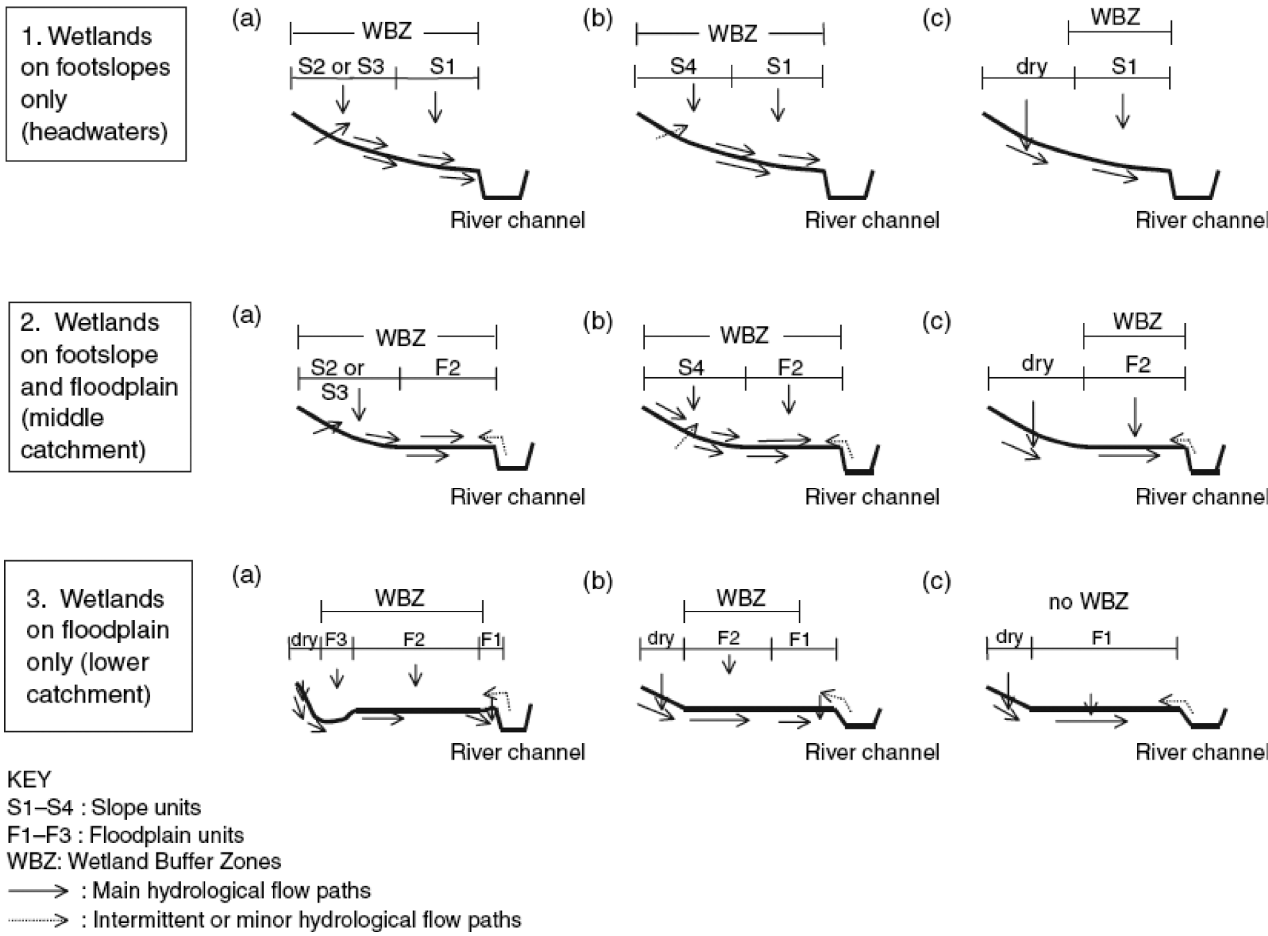

Fig. 4 Examples of generalized hydrology and geomorphic setting of river marginal wetland buffer zones and relationships to wetland functional units (Blackwell *et al.* 2009).

breaking the landscape down into features based upon their hydrology, geomorphology and soil type, and thereby would include small wetlands. By accounting for the spatial patterns and occurrence of landscape features, the user is able to assess the wetland's hydrological, biogeochemical and ecological functions qualitatively and, in some cases, quantitatively. Currently, the tool assesses 13 different functions, all of which are broken down into key processes. For example, the ES of water quality is considered in terms of nutrient retention and nutrient export, and is broken down into 24 key processes including sediment retention, denitrification and carbon storage. In addition to assessing the current ecosystem services delivered by wetlands, the tool can be used to predict the impact of changes in management, climate or other influences on the degree to which services are performed. An example of the type of outputs that can be achieved with this tool is shown in Fig. 5. This shows an area of floodplain with a small footslope wetland comprising areas of seepage, hollows and standing water, all of which are considered by this tool. The different colours reflect the degree to which the function is being performed, with explanations included in the text boxes. Tools such as this should be adapted to consider the ES delivered by wetlands (and other ecosystems) and their potential trade-offs.

CONCLUSIONS

Small wetlands can often provide important ESs, giving rise to both benefits and dis-benefits (i.e. negative impacts) that are significant across a range of scales. The extent of these services can be difficult to assess if considering small wetlands in isolation from their wider landscape context and cumulative impact in relation to other similar wetlands, as generally the former are excluded from land-use inventories due to their size. Consequently, it is important that assessments of wetland ESs do not depend upon size alone as a significant factor, but that consideration is given to other factors such as their location in the landscape

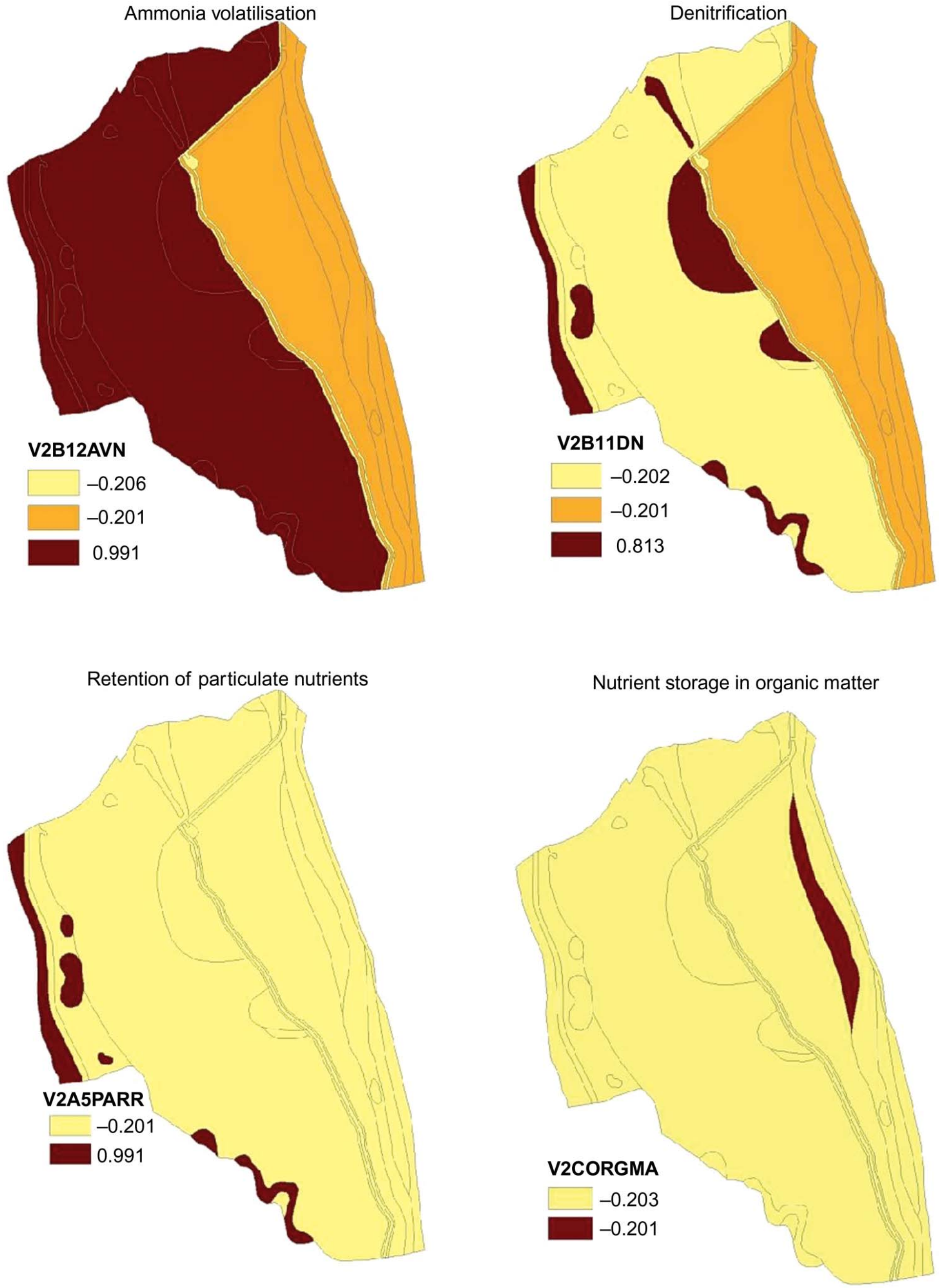

Fig. 5 Example of Wetland Evaluation Decision Support System (WEDSS) output; functional assessment of a wetland for the ecosystem service of water quality improvement via nutrient retention/removal. Individual functions of ammonia volatilization, denitrification, particulate retention and nutrient storage in organic matter are illustrated, with darker coloured areas representing higher degrees of performance.

and interaction with hydrological pathways, as many of the services result from the size of interface (e.g. buffer zones), or are inherent properties associated with their small size (e.g. nurseries for small animals).

Ultimately, there is not enough information about the distribution and functioning of the small types of wetlands we discuss here to be able to comprehensively assess the importance of the ESs they deliver. Such investigations are required to be able to understand fully their cumulative impacts on ecosystem service delivery, and also the trade-offs that arise as a result of these services, benefits and dis-benefits. Along with better understanding of these services and the processes that result in them, there is a need to develop tools which operate at the correct spatial resolution in order to include features such as small-scale wetlands. The case studies examined here show that, for some individual services, these small wetlands that are largely neglected in most wetland and land-use inventories can have significant and disproportionate impact relative to their size on the overall functioning of a landscape. Unfortunately, they often fail to be considered as a result of the scale at which classification and assessment methods are carried out. Such oversights must be addressed in future.

Acknowledgements Rothamsted Research receives grant-aided support from the UK Biotechnology and Biological Sciences Research Council. This work was funded by the Institute Strategic Programme Grant, "Ecological Processes in Multifunctional landscapes" and the Cross Institute Programme for Sustainable Soil Function, Soil CIP.

REFERENCES

Acreman, M.C. and Mccartney, M.P., 2009. Hydrological Impacts in and around wetlands. *In*: E. Maltby and T. Barker, eds., *The wetlands handbook*. First edition. Chichester: Wiley-Blackwell, 643–666.

Allan, J.D., 2004. Landscapes and riverscapes: the influence of land use on stream ecosystems. *Annual Review of Ecology, Evolution and Systematics*, 35, 257–284.

Ambus, P., 1998. Nitrous oxide production by denitrification and nitrification in temperate forest, grassland and agricultural soils. *European Journal of Soil Science*, 49, 495–502.

Baker, C., Thompson, J.R. and Simpson, M., 2009. Hydrological dynamics I: surface waters, flood and sediment dynamics. *In*: E. Maltby and T. Barker, eds., *The wetlands handbook*. First edition. Chichester: Wiley-Blackwell, 120–168.

Begg, G. 1986. The wetlands of Natal. Part 1: an overview of their extent, role and present status. Natal, South Africa: Natal Town and Regional Planning Commission, Natal Town and Regional Planning Report, Volume 68.

Beier, C. *et al*., 2008. Carbon and nitrogen cycles in European ecosystems respond differently to global warming. *Science of the Total Environment*, 407, 692–697.

Blackwell, M. and Maltby, E., 2003. Environmental and economic assessment of the location of wetland buffers in the landscape for nutrient removal from agricultural runoff. *In*: Turner, K.E., van den Bergh, J.C.J.M. and Brouwer, R., eds., *Managing wetlands: an ecological economics approach*,.Cheltenham: Edward Elgar. 165–197.

Blackwell, M.S.A., Hogan., D.V. and Maltby, E., 1999. The use of conventionally and alternatively located buffer zones for the removal of nitrate from diffuse agricultural run-off. *Water Science and Technology*, 39 (12), 157–164.

Blackwell, M.S.A., Hogan, D.V., Pinay, G. and Maltby, E., 2009. The role of buffer zones for agricultural runoff. *In*: E. Maltby and T. Barker, eds. *The wetlands handbook.* First edition. Chichester: Wiley-Blackwell, 417–439.

Blackwell, M.S.A. and Maltby, E., 2006. *Ecoflood guidelines: How to use floodplains for flood risk reduction*. Brussels: Office for Official Publicationsof the European Communities.

Blann, K.L., Anderson, J.L., Sands, G.R. and Vondracek, B., 2009. Effects of agricultural drainage on aquatic ecosystems: a review. *Critical Reviews in Environmental Science and Technology*, 39 (11), 909–1001.

Bouman, A.F., 1990. *Soils and the greenhouse effect*. Chichester: Wiley.

Bradbury, R.B. and Kirby, W.B., 2006. Farmland birds and resource protection in the UK: cross-cutting solutions for multifunctional farming? *Biological Conservation*, 129, 530–554.

Brix, H., 1994. Use of constructed wetlands in water pollution control: Historical development, present status, and future perspectives. *Water Science and Technology*, 30 (8), 209–223.

Bullock, A. and Acreman, M., 2003. The role of wetlands in the hydrological cycle. *Hydrology and Earth Systems Sciences*, 7, 358–389.

Burel, F., 1996. Hedgerows and their role in agricultural landscapes. *Critical Review in Plant Sciences*, 15, 169–190.

Cardenas, L., Rondon, A., Johansson, C. and Sanhueza, E., 1993. Effects of soil moisture, temperature and inorganic nitrogen on nitric oxide emissions from acidic tropical savannah soils. *Journal of Geophysical Research*, 98 (D8), 14783–14790.

Carey, P.D. *et al*., 2008. *Countryside survey: UK Results from 2007*. Wallingford: NERC/Centre for Ecology and Hydrology.

Cicerone, R.J., 1987. Changes in stratospheric ozone. *Science*, 237 (4810), 35–42.

Clair, T. A., Arp, P., Moore, T. R., Dalva, M. and Meng, F.-R., 2002. Gaseous carbon dioxide and methane, as well as dissolved organic carbon losses from a small temperate wetland under changing climate. *Environmental Pollution*, 116, 143–148.

Cole, S., 1998. The emergence of treatment wetlands. *Environmental Science and Technology*, 32 (9), 218–223.

Cooper, A.B., 1990. Nitrate depletion in the riparian zone and stream channel of a small headwater catchment. *Hydrobiologia*, 202 (1-2), 13–26.

Crutzen, P.J., 1970. Influence of nitrogen oxides on atmospheric ozone content. *Quarterly Journal of the Royal Meteorological Society*, 96 (408), 320.

Davies, B.R. *et al*., 2008. A comparison of the catchment sizes of rivers, streams, ponds, ditches and lakes: implications for protecting aquatic biodiversity in an agricultural landscape. *Hydrobiologia*, 597, 7–17.

Deil, U., 2005. A review on habitats, plant traits and vegetation of ephemeral wetlands—a global perspective. *Phytocoenologia*, 35, 533–705.

Dorioz, J.M., Wang, D., Poulenard, J. and Trévisan, D., 2006. The effect of grass buffer strips on phosphorus dynamics —a critical review and synthesis as a basis for application in agricultural landscapes in France. *Agriculture, Ecosystems and Environment*, 117 (1), 4–21.

Duchemin, M. and Madjoub, R., 2004. Les bandes filtrantes de la parcelle ou bassin versante. *Vecteur Environnement*, 37 (2), 36–52.

Duellman, W.E. and Trueb, L., 1986. *Biology of amphibians*. New York: McGraw-Hill.

Dunne, E.J. *et al*., 2007. Phosphorus storages in historically isolated wetland ecosystems and surrounding pasture uplands. *Ecological Engineering*, 31, 16–28.

Eglington, S.M. *et al*., 2008. Restoration of wet features for breeding waders on lowland grassland. *Journal of Applied Ecology*, 45, 305–314.

Evrard, O., Bielders, C., Vandaele, K. and van Wesemael, B., 2007. Spatial and temporal variation of muddy floods in central Belgium, off-site impacts and potential control measures. *Catena*, 70 (3), 443–454.

Freeman, C., Ostle, N. and Kang, H., 2001. An enzymic "latch" on a global carbon store—a shortage of oxygen locks up carbon in peatlands by restraining a single enzyme. *Nature*, 409 (6817), 149–149.

Grenfell, M.C., Ellery, W.N. and Preston-White, R.A., 2005. Wetlands as early warning, eco)systems water resource management. *Water SA*, 31 (4), 465–471.

Groffman, P.M., Axelrod, E.A. and Lemunyon, J.L., 1991. Denitrification in grass and forested vegetated filter strips. *Journal of Environmental Quality*, 20, 671–674.

Gustafson, A., Fleischer, S. and Joelsson, A., 2000. A catchment-oriented and cost-effective policy for water protection. *Ecological Engineering*, 14 (4), 419–427.

Hajek, M., Horsak, M., Hajkova, P. and Dite, D., 2006. Habitat diversity of central European fens in relation to environmental gradients and an effort to standardise fen terminology in ecological studies. *Perspectives in Plant Ecology, Evolution and Systematics*, 8 (2), 97–114.

Haycock, N.E. and Burt, T.P., 1993. Role of floodplain sediments in reducing the nitrate concentration of subsurface run-off: a case study in the Cotswolds. *Hydrological Processes*, 7 (3), 287–295.

Haycock, N.E. and Muscutt, A.D., 1995. Landscape management strategies for the control of diffuse pollution. *Landscape and Urban Planning*, 31, 313–321.

Heathwaite, A.L. *et al*., 2005. A tiered risk-based approach for predicting diffuse and point source phosphorus losses in agricultural areas. *Science of the Total Environment*, 344 (1-3), 225–239.

Hefting, M.M., Bobbink, R. and De Caluwe, H., 2003. Nitrous oxide emission and denitrification in chronically nitrate-loaded riparian buffer zones. *Journal of Environmental Quality*, 32 (4), 1194–1203.

Hefting, M.M., Bobbink, R. and Janssens, M.P., 2006. Spatial variation in denitrification and N_2O emission in relation to nitrate removal efficiency in a n-stressed riparian buffer zone. *Ecosystems*, 9 (4), 550–563.

Hillbricht-Ilkowska, A., 2008. The mid-European agricultural landscape: catchment-scale links between hydrology and ecology in mosaic lakeland regions. *In*: *Ecohydrology: processes, models and case studies: an approach to the sustainable management of water resources*. Wallingford UK: CAB International, 187–206.

Hogan, D.V., Blackwell, M.S.A. and Maltby, E., 2000. *Tamar 2000 SUPPORT Project Phase 2 Wetlands Report*. St Austell, UK: Report to Westcountry Rivers Trust.

Hume, C., 2008. Wetland vision technical document: overview and reporting of project philosophy and technical approach. The Wetland Vision Partnership.

Inaba, K., 1992. Quantitative assessment of natural purification in wetland for linear alkylbenzenesulfonates. *Water Research*, 26 (7), 893–898.

IPCC (Intergovernmental Panel on Climate Change), 1997. Revised guidelines for national greenhouse gas inventories *In:* J.T. Houghton, et al., eds. *IPCC guidelines for national greenhouse gas inventories*, Bracknell, UK: UK Met Office, IPCC/OECD/IES.

Johnston, C.A., 1994. Cumulative impacts to wetlands. *Wetlands*, 14 (1), 49–55.

Kadlec, R.H. and Knight, R.L., 1996. *Treatment wetlands*. Boca Raton, FL: Lewis.

Kaiser, J., 1998. New wetlands proposal draws flak. *Science*, 279, 980.

Knight, R.L., 1993. Ancillary benefits and potential problems with the use of wetlands for nonpoint source pollution control. *In*: R.K. Olson, ed. *Created and natural wetlands for controlling nonpoint source pollution*. Corvallis, OR: USEPA Environmental Research Laboratory.

Lane, C.R. and D'Amico, E., 2010. Calculating the ecosystem service of water storage in isolated wetlands using LiDAR in North Central Florida, USA. *Wetlands*, 30 (5), 967–977.

Lovett, S. and Price, P., 2006. Managing in-stream wetlands on wool producing farms. Canberra: Land and Water Australia, 8.

Maltby, E. *et al*., 2009. *Functional assessment of wetlands*. Cambridge, UK: Woodhead Publishing Ltd.

Maltby, E. and Turner, R.K., 1983. Wetlands of the world. *Geographic Magazine*, 55, 12–17.

Mander, U., Teiter, S. and Augustin, J., 2005. Emission of greenhouse gases from constructed wetlands for wastewater treatment and from riparian buffer zones. *Water Science and Technology*, 52, 167–176.

Matthews, G.V.T., 1993. *The Ramsar Convention: its history and development*. Gland, Switzerland.

Matthews, R.A. *et al*., 2010. Nitrous oxide emissions from small-scale farmland features of UK livestock farming systems. *Agriculture Ecosystems and Environment*, 136 (3-4), 192–198.

Matthews, R.A., Chadwick, D.R., Retter, A.L. and Yamulki, S., 2009. Nitrous oxide and methane emissions from small-scale farmland features in livestock systems. *16th Nitrogen Workshop*, 15–17.

McCulloch, G., Aebischer, A. and Irvine, K., 2003. Satellite tracking of flamingos in southern Africa: the importance of small wetlands for management and conservation. *Oryx*, 37, 480–483.

Millar, J.B., 1971. Shoreline area ratio as a factor in rate of water loss from small sloughs. *Journal of Hydrology*, 14 (3-4), 259–284.

MEA (Millennium Ecosystem Assessment), 2003. *Ecosystems and human well being: a framework for assessment*. Washington, DC: Island Press.

MEA, 2005a. *Ecosystems and Human Well-being : Current States and Trends*. Washington, DC; Island Press.

MEA, 2005b. *Ecosystems and Human Well-Being: Biodiversity Synthesis*. Washington, DC: World Resources Institute.

Mitsch, W.J. and Gosselink, J.G., 2007. *Wetlands*. Fourth edn. Chichester: Wiley and Sons.

Moler, P.E. and Franz, R., 1987. Wildlife values of small, isolated wetlands in the southeastern coastal plain. *In*: R.R. Odum, K.A. Riddleberger and J.C. Ozier, eds. *Proceedings of the third annual nongame and endangered wildlife symposium*, Georgia Department of Natural Resources, Atlanta, Georgia, 234–241.

Moore, M.T. *et al*., 2001. Transport and fate of atrazine and lambda-cyhalothrin in an agricultural drainage ditch in the Mississippi Delta, USA. *Agriculture, Ecosystems and Environment*, 87, 309–314.

Moron, D. *et al*., 2008. Diversity of wild bees in wet meadows:implications for conservation. *Wetlands*, 28 (4), 975–983.

Peterjohn, W.T. and Correll, D.L., 1984. Nutrient dynamics in an agricultural watershed—observations on the role of a riparian rainforest. *Ecology*, 65 (5), 1466–1475.

Petersen, S.O., Stamatiadis, S. and Christofides, C., 2004. Short-term nitrous oxide emissions from pasture soil as influenced by urea level and soil nitrate. *Plant and Soil*, 267 (1-2), 117–127.

Picard, C.R., Fraser, L.H. and Steer, D., 2005. The interacting effects of temperature and plant community type on nutrient removal in wetland microcosms. *Bioresource Technology*, 96 (9), 1039–1047.

Pilgrim, E.S. *et al*., 2010. Interactions among agricultural production and other ecosystem services delivered from European grasslands. *Advances in Agronomy*, 109, 117–154.

Potter, K.W., 1994. Estimating potential reduction flod benefits of restored wetlands. *Water Resources Update*, 97, 34–38.

Pritchard, G.C. *et al*., 2005. Emergence of fasciolosis in cattle in East Anglia. *Veterinary Record*, 157 (19), 578–582.

Ramanathan, V., 1998. Trace-gas greenhouse effect and global warming—underlying principles and outstanding issues—Volvo Environmental Prize Lecture—1997. *Ambio*, 27 (3), 187–197.

Reddy, K.R., Dangelo, E., Lindau, C. and Patrick, W.H., 1990. Urea losses in flooded soils with established oxidized and reduced soil layers. *Biology and Fertility of Soils*, 9 (4), 283–287.

Reddy, K.R. and Patrick, W.H., 1984. Nitrogen transformations and losses in flooded soils and sediments. *CRC Critical Reviews in Environmental Control*, 13 (4), 273–309.

Richter, K.O. and Azous, A.L., 1995. Amphibian occurrence and wetland characteristics in the Puget-Sound basin. *Wetlands*, 15 (3), 305–312.

Robinson, M., 1990. Impact of improved drainage on river flows. Wallingford, UK: Institute of Hydrology, Report no. 113.

Russell, K.R., Guynn, D.C., Jr and Hanlin, H.G., 2002. Importance of small isolated wetlands for herpetofaunal diversity in managed, young growth forests in the coastal plain of South Carolina. *Forest Ecology and Management*, 163, 43–59.

Rutchey, K., Schall, T. and Sklar, F., 2008. Development of vegetation maps for assessing Everglades restoration progress. *Wetlands*, 28 (3), 806–816.

Rutherford, J.C. and Nguyen M.L., 2004. Nitrate removal in riparian wetlands: interaction between surface flow and soils. *Journal of Environmental Quality*, 33, 1133–1143.

Sankowski, T.P. *et al*., 1987. The Kitsim complex, an example of a small wetland development benefitting waterfowl and cattle production. *In*: C.D.A. Rubec and R.P. Overend, eds. *Symposium '87 Wetlands/Peatlands*. Winnipeg, Canada: Ducks Unlimited Canada, 189–193.

Scott, D.A. and Jones, T.A., 1995. Classification and inventory of wetlands—a global overview. *Vegetatio*, 118 (1–2), 3–16.

Semlitsch, R.D., 2000. Size does matter: the value of small wetlands. *The National Wetlands Newsletter, January—February 2000*. Washington, DC: Environmental Law Institute, 5–13.

Semlitsch, R.D. and Bodie, J.R., 1998. Are small isolated wetlands expendable? *Conservation Biology*, 12 (5), 1129–1133.

Shilton, A., 2008. *Pond treatment technology*. London, UK: IWA Publishing.

Smart, J., Gill, J. A., Sutherland, W. and Watkinson, A.R., 2006. Grassland-breeding waders: identifying key habitat requirements for management. *Journal of Applied Ecology*, 43, 4–463.

Stevens, C.J. and Quinton, J.N., 2009. Policy implications of pollution swapping. *Physics and Chemistry of the Earth, Parts A/B/C* 34 (8-9), 589–594.

Stutter, M.I., Langan, S.I. and Lumsdon, D.G., 2009. Vegetated buffer strips can lead to increased release of phosphorus to waters: a biogeochemical assessment of the mechanisms. *Environmental Science and Technology*, 43, 1858–1863.

Thamsborg, S.M., Roepstorff, A., Nejsum, P. and Mejer, H., 2010. Alternative approaches to control of parasites in livestock: Nordic and Baltic perspectives. *Acta Veterinaria Scandinavica*, 52, (Suppl. 1), S27.

Trochlell, P. and Bernthal, T., 1998. *Small wetlands and the cumulative impacts of small wetland losses: a synopsis of the literature*. Madison, WI: Wisconsin Department of Natural Resources.

Uusi-Kamppa, J. *et al*., 2000. Buffer zones and constructed wetlands as filters for agricultural phosphorus. *Journal of Environmental Quality*, 29 (1), 151–158.

Van der Valk, A.G., Davis, C.B., Baker J.L. and Beer, C.E., 1979. Natural freshwater wetlands as nitrogen and phosphorus traps for land runoff. In: P.E. Greerson, J.R. Clark and J.E Clark, eds. *Wetland functions and values: The state of our understanding*. Minneapolis, MN: American Water Resources Association, 457–467.

Van der Zaag, A.C. *et al*., 2008. Ammonia emissions from surface flow and subsurface flow constructed wetlands treating dairy wastewater. *Journal of Environmental Quality*, 37 (6), 2028–2036.

Verhoeven, J.T.A., Arheimer, B., Yin, C. and Hefting, M.M., 2006. Regional and global concerns over wetlands and water quality. *Trends in Ecology and Evolution*, 21 (2), 96–103.

Viaud, V. *et al*., 2005. Modelling the impact of the spatial structure of a hedge network on the hydrology of a small catchment in a temperate climate. *Agricultural Water Management*, 74, 135–163.

Wang, X. *et al*., 2010. Evaluation of soil nitrogen emissions from riparian zones coupling simple process-oriented models with remote sensing data. *Science of the Total Environment*, 408, 3310–3318.

Wassen, M.J., Olde Venterink, H., Lapshina, E.D. and Tanneberger, F., 2005. Endangered plants persist under phosphorus limitation. *Nature*, 437, 547–550.

Wilbur, H.M., 1984. Complex life cycles and community organisation in amphibians. *In*: P.W. Price, C.N. Slobodchikoff and W.S. Gaud, eds. *A new ecology: novel approachesto interactive systems*. New York: John Wiley and Sons, 195–224.

Wilpiszewska I. and Kloss M., 2002. Wetland patches (potholes) in a mosaic landscape (Masurian Lakeland, Poland): floristic diversity and disturbance. *Polish Journal of Ecology*, 50, 515–525.

Wind, E. and Beese, B., 2008. Little known and little understood: development of a small wetland assessment card to identify potential breeding habitat for amphibians. *BC Journal of Ecosystems and Management*, 9, 47–49.

Wolfe, B.E., Weishampel, P.A. and Klironomos, J.N., 2006. Arbuscular mycorrhizal fungi and water table affect wetland plant community composition. *Journal of Ecology*, 94 (5), 905–914.

Yamulki, S. *et al*., 2000. Effects of dung and urine amendments on the isotopic content of N_2O released from grasslands. *Rapid Communications in Mass Spectrometry*, 14 (15), 1356–1360.

Zacharias, I., Dimitriou, E. and Koussouris, T., 2003. Developing sustainable water management scenarios by using thorough hydrologic analysis and environmental criteria. *Journal of Environmental Management*, 69 (4), 401–412.

Zaman, M. *et al*., 2008. Nitrous oxide generation, denitrification, and nitrate removal in a seepage wetland intercepting surface and subsurface flows from a grazed dairy catchment. *Australian Journal of Soil Research*, 46, 565–577.

Zierholz, C., Prosser, I.P., Fogarty, P.J. and Rustomji, P., 2001. In-stream wetlands and their significance for channel filling and the catchment sediment budget, Jugiong Creek, New South Wales. *Geomorphology*, 38, 221–235.