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Ecosystem services delivered by small-scale wetlands

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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. À 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 livrés 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...
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.

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 \text{ 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 \text{ 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-km² area. More than 46% of the wetlands were less than 1.2 ha, although this is almost certainly an underestimate 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).

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

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

### 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).

<table>
<thead>
<tr>
<th>Service</th>
<th>Description</th>
<th>Examples</th>
</tr>
</thead>
<tbody>
<tr>
<td>Agricultural production</td>
<td>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.</td>
<td>Food (meat, milk), fibre (e.g. cotton), fuel (e.g. wood, biofuel).</td>
</tr>
<tr>
<td>Water quality</td>
<td>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 transfer 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.</td>
<td>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.</td>
</tr>
<tr>
<td>Hydrological regulation</td>
<td>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).</td>
<td>Storage of water during rainfall events can help alleviate downstream flooding.</td>
</tr>
<tr>
<td>Biodiversity conservation</td>
<td>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).</td>
<td>Wetlands such as calcareous fens provide habitats for specialized and rare plants and animals.</td>
</tr>
</tbody>
</table>
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 (Rutcheay 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$_2$O, 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...
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 Beece 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 (CH4) and nitrous oxide (N2O). 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 (NH3), 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$_2$O), 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

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**Fig. 3** Trade-offs between the deliveries of multiple 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$_2$O 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$_2$O 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$_2$O 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$_2$O 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$_2$O were emitted from small-scale farmland features compared with adjacent pasture land. Overall, the greatest annual N$_2$O 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$_2$O 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$_2$O 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$_2$O 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$_2$O fluxes from farms. Currently, many of these sources are not considered by the Intergovernmental Panel on Climate Change methodology (IPCC 1997) for determining N$_2$O 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$_2$O 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).

<table>
<thead>
<tr>
<th>Farm</th>
<th>$\text{t CH}_4$ per farm</th>
<th>Produced as by-product of enteric fermentation</th>
<th>Emissions from animal manures</th>
<th>Total farm emission (IPCC)</th>
<th>Total emission from small-scale features</th>
</tr>
</thead>
<tbody>
<tr>
<td>Farm I</td>
<td>21.42</td>
<td>3.90</td>
<td>25.32</td>
<td>22.06</td>
<td></td>
</tr>
<tr>
<td>Farm II</td>
<td>16.52</td>
<td>3.40</td>
<td>19.98</td>
<td>2.83</td>
<td></td>
</tr>
<tr>
<td>Farm III</td>
<td>20.36</td>
<td>1.64</td>
<td>21.99</td>
<td>3.71</td>
<td></td>
</tr>
<tr>
<td>Farm IV</td>
<td>6.29</td>
<td>0.64</td>
<td>6.92</td>
<td>10.92</td>
<td></td>
</tr>
</tbody>
</table>

Table 4 shows the 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). The table includes the total emissions from enteric fermentation, animal manures, and small-scale features on each farm.

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$_2$O than field margins, whilst wetland forested buffers produce seven times more N$_2$O 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
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

**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.
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.

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