Biodivers Conserv (2011) 20:2461–2481
DOI 10.1007/s10531-011-9998-y

ORIGINAL PAPER

Impacts of species-led conservation on ecosystem services of wetlands: understanding co-benefits and tradeoffs

Brendan Fisher · Richard B. Bradbury · Julian E. Andrews · Malcolm Ausden · Stephanie Bentham-Green · Sue M. White · Jennifer A. Gill

Received: 9 August 2010 / Accepted: 21 January 2011 / Published online: 8 February 2011

Abstract Biodiversity conservation organisations have recently begun to consider a wider ecosystem services context for their activities. While the literature suggests the potential of 'win–win' situations where biodiversity conservation and the delivery of ecosystem services overlap, empirical evidence is wanting. Here we explore the role that species-led management for the benefit of biodiversity in cultural landscapes can play in the delivery of wider ecosystem services. We use UK lowland wetlands as a case study and show how successful delivery of species-led conservation through management interventions relies on practices that can affect greenhouse gas fluxes, water quality and regulation, and cultural benefits. In these wetlands, livestock grazing has potentially large effects on water and greenhouse gas related services, but there is little scope to alter management without compromising species objectives. Likewise, there is little potential to alter reedbed management without compromising conservation objectives. There is some potential to alter woodland and scrub management, but this would likely have limited influence due to the relatively small area over which such management is practiced. The management of water levels potentially has large effects on provision of several services and there does appear to be some scope to align this objective with biodiversity objectives.

B. Fisher and R. B. Bradbury are contributed equally to this work.

B. Fisher (✉)
Woodrow Wilson School of Public and International Affairs, Princeton University, Princeton, NJ 08544, USA
e-mail: bpfisher@princeton.edu

R. B. Bradbury · M. Ausden
Royal Society for the Protection of Birds, Sandy SG19 2DL, UK

B. Fisher · J. E. Andrews · S. Bentham-Green
School of Environmental Sciences, University of East Anglia, Norwich NR4 7TJ, UK

S. M. White
School of Applied Sciences, Cranfield University, Bedford, UK

J. A. Gill
School of Biological Sciences, University of East Anglia, Norwich NR4 7TJ, UK

A comprehensive understanding of the *net* costs and benefits to society of these interventions will require fine-grained research integrating ecological, economic and social science research. However, a less analytic understanding of the potential costs and benefits can highlight ways by which land management principally to achieve biodiversity conservation objectives might be modified to enhance delivery of other ecosystem services.

Keywords Ecosystem services · Biodiversity · Greenhouse gas flux · Climate change mitigation · Water regulation · Reedbed · Lowland wet grassland · Ecosystem approach · Wetlands · Conservation

Introduction

While many targeted biodiversity conservation initiatives have been successful at conserving species and habitats (Rodrigues 2006; Wilcove 2008), the high rate of global species decline and loss (Butchart et al. 2010) demonstrates that biodiversity conservation often loses out to other interests (Wilcove et al. 1998). The intense pressure created by land-use demands from humans for water, agricultural production, other natural resources and habitation is likely to make biodiversity conservation per se increasingly difficult (Millennium Assessment 2005). Meanwhile, *ecosystem services* has become an umbrella concept for linking biodiversity and the functioning of ecosystems to human welfare (Millennium Assessment 2005; Daily et al. 2009). However, a practical question remains: could biodiversity conservation be made more resilient by making the connections between ecosystem function, ecosystem services and human welfare more fully understood and recognised in decision-making? This question provides challenges for policy makers, academics and conservation organisations alike (e.g. Defra 2007; Redford and Adams 2009; RSPB 2009; Harrison 2010).

Slowly an evidence base of the importance of ecosystem services is being built. For example, a small body of evidence suggests that protecting 'natural' habitats provides a net economic gain to society compared to their alternative converted states (Balmford et al. 2002; Turner et al. 2003; Andam et al. 2010) (Fig. 1a), and that ecological restoration increases provision of both biodiversity and ecosystem services, but that values of both remain lower in restored than intact reference ecosystems (Rey Benayas et al. 2009). Several other studies have suggested limited congruence between places important for biodiversity conservation and those important for provision of services (Naidoo et al. 2008; Anderson et al. 2009; Egoh et al. 2009; Nelson et al. 2009). However, the delivery of services in protected areas is also likely to vary depending on the land management undertaken for conservation in these areas (Fig. 1b, c).

Understanding the variation in service delivery resulting from different management practices is clearly important from a practitioner and policy perspective. This is especially true in highly modified 'cultural landscapes', such as most of lowland Europe, where intervention management is often used to prevent succession to benefit particular habitats, species or assemblages of species. How could such conservation efforts be adapted to enhance results for a range of other societal goods and services (Fig. 1c)? Aligning such species-led conservation with a wider ecosystem service agenda requires a systematic understanding of management activities and their implications for relevant ecosystem services. This understanding helps to identify the synergies and trade-offs between species conservation and ecosystem services provision, and consequent monitoring and research priorities.

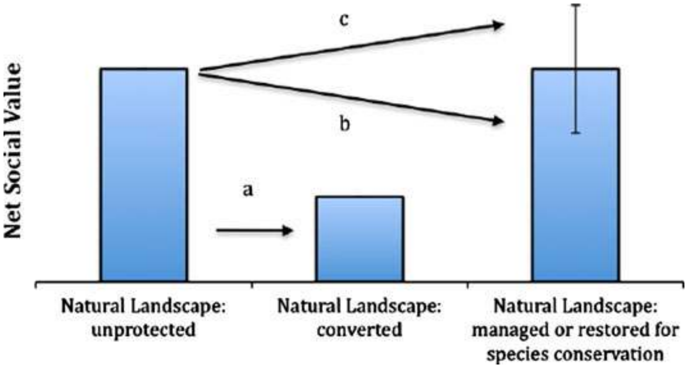

Fig. 1 A schematic representation of how the net social value of an ecosystem might change between different states; **a** conversion of wild nature to typical alternative land-use (e.g. development, agriculture), **b**, **c** management of the ecosystem to deliver particular species outcomes showing potential to decrease (**b**) or increase (**c**) the value of the system, attributable to impacts on services of intervention management for species-led conservation purposes

In this paper, we examine this issue in the context of UK lowland wetlands, specifically the portfolio of lowland wet grassland and reedbed reserves managed by the Royal Society for the Protection of Birds (RSPB) in Southeast and Eastern England. Given increasing scarcity of water resources, the importance of managing seasonal high flows across a landscape (e.g. floods) and concerns over food security, the long-term sustainability of wetland conservation may depend upon more than just their benefits for biodiversity (Posthumus et al. 2010). While we know that wetlands benefit human society through the provision of a wide range of ecosystem services (Turner et al. 2000; Andrews et al. 2006; Turner and Daily 2008), little is known about the consequences of species-led wetland management for wider ecosystem service provision.

Here we explore whether conservation activities in these landscapes deliver on their stated conservation objectives, and how specific management interventions undertaken to deliver biodiversity and recreational opportunities may influence and interact with other ecosystem services. In doing so, we introduce an approach which could be used by protected area managers to understand and assess the effects of species-led conservation on other ecosystem services.

Methods

Study sites

Royal Society for the Protection of Birds landholdings in the UK comprise 206 reserves, covering 142,000 hectares. Wetland habitats make up a large proportion of this land, with 35,000 hectares in 156 reserves spread across the UK. Approximately one-third of this wetland area is in the East and Southeast regions of England, and 67% of the total lowland wet grassland area on RSPB reserves is located in these regions. We selected all 22 reserves within these regions that contain managed freshwater habitats (i.e. lowland wet grassland, reedbed, open water, fen, scrub and wet woodland). Estuarine habitats within these sites were excluded from analyses.

The 22 reserves comprised 4,471 ha of lowland wet grassland, 838 ha of reedbed and fen, 304 ha of scrub and wet woodland, 291 ha of open water, and 37 ha of tall–herb-dominated vegetation. Of the lowland wet grassland, 1,890 ha (12 sites) contain coastal grazing marsh (grassland on reclaimed saltmarsh), 731 ha (4 sites) floodplain grassland (lying in the floodplain of rivers), and 1,850 ha (2 sites) washlands. Washlands are artificial floodplains constructed from the Sixteenth Century onwards to aid land drainage in lowland areas. They consist of a flat, embanked area lying between a river and artificial relief channel, or between two artificial relief channels, into which water is diverted and stored during periods of high flow, to reduce flooding in other parts of the catchment (English Nature 2001).

Identifying the aims of conservation management, and management activities undertaken for species-led conservation

Each RSPB nature reserve has a Management Plan, which contains quantified targets for the condition of species, assemblages of species, and habitats at the site. The objectives in each Management Plan reflect the conservation priorities of the RSPB and, at sites with national or international conservation designations, also statutory conservation objectives. Of the 22 focal reserves, 20 were all or part Sites of Special Scientific Interest (SSSIs), 14 were all or part Special Protection Areas (SPAs), and two were all or part Special Areas of Conservation (SACs). We collated information from the conservation objectives in these Management Plans to describe the species that intervention management aims to benefit in each of the main wetland habitats at these sites.

Site Managers at the reserves compile Annual Reports, which describe the management carried out on their reserve. We used the 2008–2009 Annual Reports of the 22 focal reserves to collate (a) the conservation management actions carried out in each of the main habitats at each site, and (b) the area over which each action took place.

Monitoring the impacts of management on species of conservation concern

We measured the success of conservation management by assessing whether the population trends of the principal breeding bird species of high conservation value at these sites were favourable, particularly in relation to national population trends.

We used annual count data from 1990 to 2009 for the target species on reserves, which are monitored using standard methodologies (O'Brien and Smith, 1992; Gilbert et al. 1998), to determine population trends of these species across RSPB reserves in our case study area. During this period, several new blocks of wet grassland or arable land (for conversion to wet grassland) were acquired by RSPB, allowing us to assess changes in population trends of target species on these blocks of land following their acquisition and management as nature reserves. We collated changes in numbers of breeding waders for the nine blocks of land (total area = 1,287 ha) acquired at the 22 sites that had been managed as nature reserves for 9 years or more. (Four other blocks were managed for an average of 5 years and show similar trends but were not included, as we assumed that perhaps management interventions are not responsible for trends over this shorter period). We also assessed the success of reedbed management for target species by collating numbers across the entire area of reedbed present on RSPB reserves in E and SE England during the period 1990–2009, and on one site at which new reedbeds had been created (Lakenheath Fen: 104 ha of reedbed).

Identifying other ecosystem services associated with conservation management activities on wetlands

We consulted fifteen experts and stakeholders from a range of UK government, NGO and academic institutes with interests in wetland conservation, management and research, to explore the range of management activities currently employed on wetlands in the UK and their potential to provide ecosystem services and disservices. We hosted one large workshop with all fifteen experts, and then followed that meeting up with two smaller meetings, with participants selected to help elaborate specific details of management impacts on hydrology and greenhouse gas fluxes respectively. The follow-up meetings were structured to answer specific questions raised in the large workshop, discuss relevant literature and investigate data sources. We also held a further meeting to explore the availability of information on cultural services.

These cultural services are delivered in addition to conserving species, since wetlands also provide opportunities for recreational activities such as walking and bird-watching. The RSPB reserves aim to deliver their conservation objectives in ways that produce wider benefits, including public enjoyment and well-being, formal and informal education, and contributions to local economies. To understand whether wetland management activities undertaken to deliver conservation goals influence the provision of cultural services, we explored the motivations of visitors to these reserves by analysing questionnaire surveys carried out by the RSPB. On six of our 22 reserves, visitor surveys were carried out between October 2007 and September 2008 (total number of respondents = 2,054). Questions related to motivations for visits and distances travelled to sites.

Results

Aims of conservation management, and wetland management activities undertaken

The lowland wet grassland, reedbeds and fens at the 22 reserves are managed to benefit a range of species considered characteristic of these habitats. Particular priority is given to breeding waders and wintering wildfowl on lowland wet grassland, and breeding Great Bitterns *Botaurus stellaris* in reedbeds (Table 1). The suite of breeding waders comprises Northern Lapwing *Vanellus vanellus*, Common Redshank *Tringa totanus* and Common Snipe *Gallinago gallinago*. These species are conservation priorities as a consequence of severe population declines of breeding waders in the UK, principally resulting from agricultural intensification and drainage of wet grassland (Smart et al. 2006, 2008; Eglington et al. 2009). The very low numbers of Great Bitterns are a result of changes in the extent and condition of reedbeds (Tyler et al. 1998; Wilson et al. 2005).

The main conservation management activities on lowland wet grasslands are control of water levels, vegetation management through livestock grazing and cutting, rotational cleaning out of water-filled ditches, and excavation and maintenance of other shallow water bodies (Table 2). Water-table levels are kept high in winter, spring and early summer, principally to provide areas of shallow surface water for wildfowl in winter, and high field water levels and patchy, shallow water for waders to feed in during their breeding season. Water-table levels are controlled by manipulating ditch water levels with gravity-driven control structures, and in some cases by pumping. Shallow water bodies are created in grassland principally to provide areas favoured by feeding waders.

Table 1 Features (species and assemblages of species) referred to in management plan objectives for areas of lowland wet grassland and reedbed/fen on RSPB reserves in Eastern and South-eastern England

Features referred to in reserve management plan objectives	Number of sites which refer to the feature in their management plan objectives
(a) Lowland wet grassland (18 sites)	
Breeding waders	18
Wintering wildfowl	13
Water vole *Arvicola terrestris*	9
Ditch flora	9
Ditch invertebrate fauna	8
Breeding wildfowl	5
Specific plant species (non-ditch)	5
Specific breeding passerine species	4
Breeding spotted crake *Porzana porzana*	3
Wintering birds of prey	3
Common crane *Grus grus*	2
Breeding corncrake *Crex crex*	1
(b) Reedbed/fen (17 sites)	
Specific breeding passerine species	11
Breeding bittern *Botaurus stellaris*	9
Water vole *Arvicola terrestris*	7
Specific moth species	7
Specific invertebrate species (other than moths)	5
Fen plant communities	5
Specific plant species	4
Otter *Lutra lutra*	3
Breeding water rail *Rallus aquaticus*	3
Wintering waterfowl	2
Breeding wildfowl	2
Breeding marsh harrier *Circus aeruginosus*	2
Amphibian species	2
Breeding Common Snipe *Gallinago gallinago*	1
Breeding common crane *Grus grus*	1
Black poplar *Populus nigra*	1

Most vegetation management was by summer grazing (all sites), with mechanical ‘topping’ carried out over approximately half of the grazed area to remove tall stems and undesired ‘weed’ species avoided by livestock. Only very small areas of the grassland are left unmanaged in any given year, principally to provide taller vegetation adjacent to water-filled ditches for Water Voles *Arvicola terrestris*, and to encourage small mammal populations.

The livestock grazing on these lowland wet grasslands is primarily commercial beef cattle from local graziers. Cattle comprised $\sim 80\%$ of the total livestock numbers on the 22 reserves, and $\sim 94\%$ of the total grazing pressure (calculated from standard livestock units

Table 2 The number of RSPB reserves in E and SE England at which different management techniques are used, and the area of managed (a) grassland and (b) reedbed/fen in 2008/2009

Management technique	Number of sites	Area of land managed in 2008/2009	
		% of total	Length (m/ha)
(a) Lowland wet grassland (18 sites)			
Sward management			
Grazing only	18	42.3	
Grazing and topping	17	46.1	
Grazing and weed-wiping with herbicide	12	3.0	
Grazing and hand pulling of 'weed' species	3	1.8	
Mowing and aftermath grazing	11	6.4	
Fertiliser application	1	1.6	
Muck spreading	3	–	
Sub-soiling	3	0.3	
Chisel ploughing	1	–	
Rotovating	4	0.01	
Chain harrowing	1	–	
Rolling	1	–	
Total area of sward managed	–	98.6	
Management of trees and scrub			
Tree removal/pollarding/coppicing	6	Not measured	
Water level management			
Control of water levels	18	100.0	
Management of water bodies			
Excavation of scrapes	7	0.004	
Clearance of vegetation from pools	1	0.003	
Excavation/maintenance of footdrains	5		>0.6
Excavation of ditches	5		0.4
Cleaning of ditches	16		13.7
Bund creation	2		0.2
(b) Reedbed/fen (17 sites)			
Management of swamp/fen vegetation			
Winter cutting	12	1.4	
Summer cutting	6	0.7	
Grazing only	4	9.2	
Grazing and cutting	1	0.5	
Burning	2	<0.1	
Total area of vegetation managed	–	11.8	
Scrub management			
Removal of scrub	9	2.3	
Water level management			
Control of water levels	17	100.0	
Management of water bodies			
Excavation of pools	10	0.18	

Table 2 continued

Management technique	Number of sites	Area of land managed in 2008/2009	
		% of total	Length (m/ha)
Clearance of vegetation from pools	6	0.16	
Excavation/cleaning of ditches	10		11.1

for different classes of livestock). Sheep were also commonly used, particularly during winter, and comprised ~18% of the total livestock numbers and ~4% of the maximum grazing pressure. Ponies were only used at one site, and comprised only ~2% of both livestock numbers and grazing pressure.

Grazing is carried out at low levels in spring (usually commencing in May) and early summer to reduce the risk of wader nests being trampled (Beintema and Müskens 1987), and at higher levels in summer and autumn (usually until October or November) to attain suitable sward conditions for waterfowl in winter and breeding waders the following spring. The mean maximum grazing pressure was 1.6 livestock units per ha. Low intensities of grazing also took place in winter at eight sites. Cattle are preferred to other livestock because they create the more heterogeneous sward favoured by breeding waders and more varied vegetation structure along ditch edges, and because cattle trample fewer nests than sheep, per quantity of vegetation removed. Water level control is also important for grazing, as it is important to limit the extent to which livestock feet get wet.

Mowing is carried out over a small proportion of the grassland, principally to maintain the characteristic flora of agriculturally unimproved hay meadows and, at one site, to provide suitable habitat for breeding Corncrakes *Crex crex* (e.g. Green 1996). Management of problem plant species includes weed-wiping with herbicide of *Juncus* spp. and *Cirsium* spp. and hand-pulling of Ragwort *Senecio jacobaea*. Mechanical rotovation is occasionally used to improve the structure of compacted soil, particularly on grassland created on ex-arable land.

Rotational cleaning of water-filled ditches is undertaken to maintain suitable conditions for a wide range of wetland plants and invertebrates, to facilitate the transport of water around sites, and to act as barriers to the movement of livestock between fields. The material removed from ditches is sometimes rotovated to break up clods. At some sites, colonising scrub is removed and trees and scrub are occasionally pollarded and coppiced to remove potential perches and nest sites for predatory birds (e.g. van der Vliet et al. 2008).

Management of reedbeds and fens also involves control of water levels and the maintenance and creation of open water in ditches and pools. Water level management in reedbeds aims to maintain standing water in a proportion of the reedbed during the spring and summer to provide suitable foraging conditions for breeding Great Bitterns (Gilbert et al. 2005, 2007), and to provide a range of successional stages from open water to dry reedbed and fen and scattered scrub for other wetland species (See Table 1).

In contrast to lowland wet grassland, only a small proportion of the vegetation in reedbeds and fens is managed each year. Cutting of small sections of reedbed in winter aims to prevent accumulation of litter, while cutting of patches of fen in summer encourages high plant species-richness. Commercial cutting of reed for thatching only took place at two sites, and commercial summer cutting of fens for *Cladium mariscus* for thatching at one site. Material that was not cut commercially was usually burnt on site, with a smaller proportion piled up and left. Grazing of reedbeds and fens to provide a mosaic of vegetation types and structures was carried out during both summer and winter at four sites and in summer at one site, by cattle (2 sites), cattle and sheep (1 site), cattle and ponies (1 site), and ponies (1 site). The mean maximum grazing intensity was 0.5 livestock units per ha, using only Highland Cattle and Konik ponies, which are both hardy, non-commercial breeds well-suited to being out-wintered and surviving on coarse vegetation. All grazing of reedbeds and fens was managed on a non-commercial basis.

Intervention management of wet woodland consisted of coppicing on 10 sites; pollarding of trees on 2 sites; thinning of trees on 1 site; removal of non-native tree species on 1 site; and planting of 28 ha of hybrid poplars *Populus* spp. to provide habitat for Eurasian Golden Orioles *Oriolus oriolus* (1 site). No sites manage water levels within these woodlands, although they can be influenced by water level management in adjacent wetland habitats.

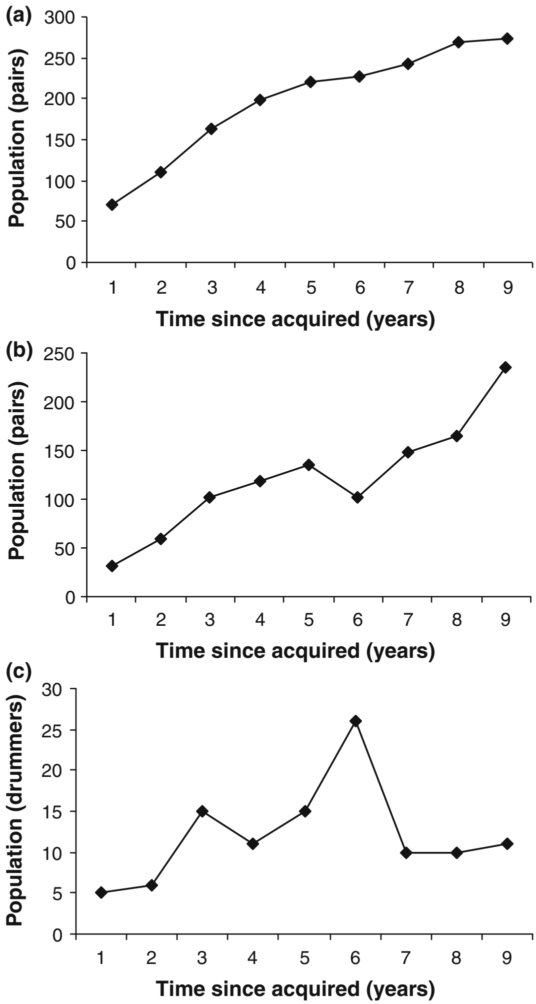

Fig. 2 Changes in total numbers of breeding **a** Northern Lapwing, **b** Common Redshank and **c** Common Snipe on nine areas of land (total area = 1,287 ha) in Eastern and Southeast England in relation to the time since they were acquired by RSPB. All sites were acquired since 1990 and have been managed as nature reserves for ≥9 years

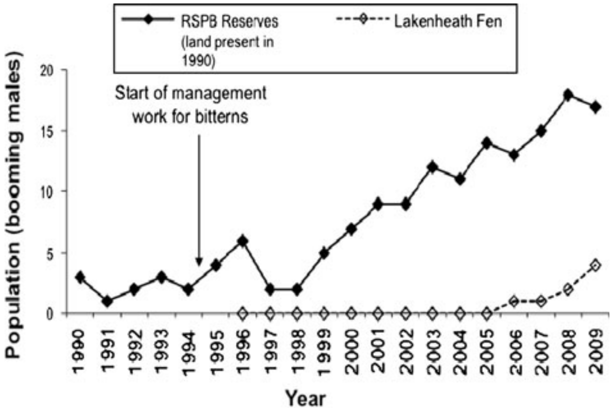

Fig. 3 Changes in numbers of booming Great Bitterns on the area of RSPB reserves present in Southeast and Eastern England in 1990 (five sites comprising 218 ha of reedbed) and at Lakenheath Fen, 242 ha of arable land acquired in 1995 on which reedbed and other wetland habitats have subsequently been created

Impacts of management on key species

Populations of Northern Lapwings, Common Redshank and Common Snipe all increased on land acquired and restored to wet grassland on the 22 focal reserves (Fig. 2a–c). These population increases are in contrast to large-scale declines that have taken place on lowland wet grassland elsewhere in lowland England over this period (Wilson et al. 2005, 2007).

Numbers of Great Bitterns recorded at the focal reserves also increased since the late 1990s (Fig. 3), in line with increases elsewhere in Britain that have been attributed to conservation management of reedbeds on nature reserves (Wotton et al. 2009).

On land for which RSPB is responsible for delivering statutory favourable conservation status, government indicators also show that the objectives for each relevant feature are either being met or being moved towards (English Nature 2010). Thus, the management of RSPB reserves appears to be effective at meeting species-led conservation objectives.

Identifying potential impacts of conservation management activities on ecosystem services

The workshop and follow-up interviews identified three categories of key ecosystem services and benefits relevant to the management of UK lowland wetlands: hydrological (water flow and quality), greenhouse gas flux, and cultural/socio-economic services. These three broad categories were then linked to the range of management activities on the focal reserves, in order to consider how each activity could influence the delivery of associated services and benefits as well as the costs (or disservices) that management might impose (Table 3). Here we report the relationship between management activities and service provision via the service categories in order to highlight the largest potential impacts.

Hydrological services and disservices

Water flow on wetlands can be influenced by the main management activities undertaken in all four major habitats (Table 3). The control over water levels that is carried out on all wet grasslands on these reserves (Table 2) can influence flood storage capacity and

Table 3 Potential services and disservices associated with specific management activities for biodiversity conservation on the four main lowland wetland habitats (context-specific issues are highlighted in italics, see text for details)

Habitats	Service and benefit categories							
	Water flow		Water quality		Greenhouse gas flux		Cultural/socio-economic	
	Services	Disservices	Services	Disservices	Services	Disservices	Services	Disservices
Grassland								
Grazing		Soil compaction influence on water infiltration & risk of flooding		Potential nutrient & faecal coliform contamination of water courses or groundwater	Compaction of waterlogged soils may facilitate carbon storage	Compaction of waterlogged soils may increase denitrification; Methane output from cattle and water logged soil	Maintaining traditional rural landscape; Opportunities for local graziers	
Topping/ mowing		Soil compaction influence on water transition & risk of flooding				Potential small amount of methane emission	Maintaining traditional rural landscape; Local employment: Provision of hay	
Problem species control						Burning removed material may slightly increase in carbon emissions	Influence on local landscape; Reduced need for control on adjacent land	
Rotovating	May increase water infiltration			May increase sediment influx into waterbodies		May slightly increase carbon emission (esp. in organic soils)		

Table 3 continued

Habitats	Service and benefit categories							
	Water flow		Water quality		Greenhouse gas flux		Cultural/socio-economic	
	Services	Disservices	Services	Disservices	Services	Disservices	Services	Disservices
Open water								
Water level control	*Influences flood storage potential*; May influence groundwater recharge	*Influences flood storage potential*; May influence groundwater recharge			Carbon storage can be high in waterlogged soils	Dentitrification and methanogenesis can be high in waterlogged soils	Influence on local landscape; *May influence local flood potential*	*May influence local flood potential*
Wet feature creation	Increased area for water storage	May influence local flood potential				May slightly increase denitrification	Influence on local landscape	
Ditch reprofiling & de-silting	Increased area for water storage	May influence rate of water transfer & local flood potential				Potential slight increase in rate of carbon emission		
Reedbed								
New reedbed planting	*Can influence evapo-transpiration rate*; May influence flood storage	*Can influence evapo-transpiration rate*	May improve water quality			Reeds may provide routes for greenhouse gas transport	Influence on local landscape	

Table 3 continued

Habitats	Service and benefit categories							
	Water flow		Water quality		Greenhouse gas flux		Cultural/socio-economic	
	Services	Disservices	Services	Disservices	Services	Disservices	Services	Disservices
Reedbed cutting	Can influence evapo-transpiration rate				Lowered water levels may reduce denitrification; Emissions from use of cut reed; Potential biofuel	Lowered water levels may reduce carbon storage; Emissions from use of cut reed	Maintaining traditional cultural activities; Provision of reed for thatching	
New channel creation	Increased area for water storage	May influence local flood potential						
Scrub & woodland								
Scrub removal/ tree felling	Reduced water use by trees may increase water resource	Reduced regulation of water flow by trees may increase flash flood risk			Felled trees buried in waterlogged soils may increase carbon storage; Potential biofuel		Influence on local landscape; Provides materials for local use	
Tree planting	Water flow regulation by trees may reduce flash flood risk	Increased water use by trees may decrease water resource			Increased carbon storage		Influence on local landscape	
Coppicing					Potential biofuel		Provide materials for local use	

groundwater recharge. However, whether this capacity acts as a service or a disservice will depend on several context-specific factors (Bullock and Acreman 2003). Impacts of water control on flooding will depend on the water levels that are maintained on each site, the timing of high water level relative to timing of high flows, and the potential available volume of storage on each site relative to other flood storage in the catchment. Here, there is a clear potential trade-off on lowland wet grasslands between maintaining high water levels to provide suitable conditions for wintering wildfowl and breeding waders, and for the wetland fauna and flora as a whole, and maintaining low water levels to maximise flood storage capacity (Ratcliffe et al. 2005). Flood storage here will only realise significant ecosystem service benefits if it helps to prevent damage to life, property, and agricultural interests. Typically, in the UK, where water levels on reserves are held high in winter and spring this will coincide with the time of greatest need for flood storage, thus the best-case scenario might be no net-benefit. Groundwater recharge is likely to be enhanced where water levels are held high, and will be influenced by the duration of high water level, the depth of water above the soil surface and the hydraulic conductivity of the soil and soil water saturation at each location.

The creation of wet features and reprofiling of ditches in grasslands or reedbeds can increase the water storage capacity of sites, but if high water levels are maintained in these features then local flood risks will not be reduced. In addition, cleared ditches may accelerate water movement away from the site, potentially adding to local flood risk.

Planting and cutting of reedbeds can affect evapo-transpiration rates, removing water from the catchment system and thus impacting on water resources, although this effect can vary depending on the size of the reedbed (Peacock and Hess 2004). In general, smaller reedbeds, which are the norm in the UK, have been found to have higher evapotranspiration rates due to advective effects relating to the surrounding land uses. It is important to consider how the evapotranspiration rate of reed-beds relates to any vegetation it is replacing. Creation of wet woodland, too, will increase evapotranspiration rates. This, and the associated increase in surface roughness, will impact on water flows.

Water flow and regulation can also be influenced by the removal of trees and scrub, changing the water balance components. Depending on the magnitude of the biomass removal, this could be a service in areas where more water is needed, or a disservice in areas where flooding is an issue. Soil compaction by livestock can potentially increase the proportion of water transition by rapid surface routes, and consequent flood risk (Holman et al. 2003; O'Connell et al. 2004; Hess et al. 2010). It may also affect downstream water quality, if increased surface flow heightens pollution risk. There will be a concomitant reduction in groundwater recharge under compacted soils. On the RSPB reserves, site managers avoid heavy grazing in spring when soils are wet and vulnerable to compaction, not least to minimise levels of nest trampling. However, the winter grazing required to achieve suitable sward conditions for breeding waders the following spring always has the potential to cause soil compaction. Livestock grazing can also potentially result in nutrient or faecal coliform contamination of water bodies (Ferguson et al. 2003; Withers and Sharpley 2008; Jarvie et al. 2008) although, again, under reserve management intensities this effect might be marginal.

Reed-beds are increasingly used as constructed natural water treatment methods (Brix et al. 2007; Vymazal 2007; Cooper 2009), due to their uptake of nutrients and trapping of sediments. Again, it is important to consider the nutrient uptake relative to any vegetation that the reed-beds are replacing. Reed-bed systems have a finite nutrient trapping life and, unless biomass is harvested (thus removing nutrients from the site), capacity to trap pollutants can be exceeded resulting in higher nutrient and sediment losses from reed-beds during storms.

Greenhouse gas flux services and disservices

All the management activities carried out on these wetlands have the potential to influence climate change mitigation. This is via both carbon storage, through burial of organic matter, and greenhouse gas emissions, principally the production of methane (CH_4) and nitrous oxide (N_2O) (Table 3). Compaction of soils by livestock, maintaining waterlogged soils and, burying felled trees in waterlogged soils can all reduce emissions of carbon dioxide (CO_2) from organic soils in some situations (Dobbie and Smith 2003; Smith et al. 2003). These activities also help to increase carbon (organic matter) storage within flooded soil (Cook 2007). This is mainly because the effects of relatively efficient aerobic bacterial metabolisms on organic matter degradation are decreased by water logging, being replaced by less efficient anaerobic microbial consortia, principally nitrate-reducers and fermenters. However, CH_4 production may be increased either directly from enteric fermentation by livestock, or from bacterial fermentation in waterlogged soils: wet and compacted soils will also typically increase rates of soil-zone denitrification and hence N_2O emissions. These emissions are important as they have significant global warming potential (GWP)—on a 100 year time-frame, CH_4 has a 21 times greater GWP than CO_2, while N_2O has a 310 times greater GWP than CO_2 (Houghton et al. 1995).

The creation and maintenance of reedbeds can increase carbon storage, but with the potential disservice of increasing both de-nitrification and methane production by linking the soil zone with the atmosphere. Terrestrial swamp and marsh environments are significant global sources of biogenic methane (Bartlett and Harriss 1993; Whiting and Chanton 2001). Burning of cut vegetation immediately emits the CO_2 that has been sequestered into the plant, though reedbed cutting, scrub removal and coppicing all have the potential to provide materials for bioenergy, which can offset some CO_2 emissions from fossil fuels. The greenhouse gas flux consequences of wetland management will therefore depend greatly on the subsequent treatment and uses of the cut vegetation.

Cultural and socio-economic services and disservices

Species-led interventions both directly and indirectly affect cultural and economic outcomes. Management interventions by their nature can provide cultural services, as in the maintenance of traditional cultural activities such as reed cutting and reedbed management, along with the use of cut reed for thatching or fuel purposes. Also, summer grazing by cattle and sheep for meat production is a key management technique used to achieve biodiversity conservation objectives on lowland wet grasslands, as is year-round grazing by cattle and ponies managed on a non-commercial basis to achieve biodiversity conservation objectives in reedbeds and fens (Table 2). These interventions directly maintain cultural activities and traditional rural landscapes. Additionally, this type of low-input grazing system provides income opportunities for local graziers (Table 3).

One beneficial feature of washlands and inundated floodplain grasslands is that they make use of nutrients in sediment deposited by floodwater, which would otherwise be lost from agricultural systems. Nutrients deposited in sediment increase livestock production, with no need for additional manufacture of fertiliser, and its associated ecosystem service dis-benefits. Productivity of washland vegetation is high, with dry matter yield of washland vegetation dominated by grasses and sedges being similar to that of agriculturally improved grassland (Morris et al. 2008).

In addition to the direct services provided by the management interventions themselves, the outcomes of these interventions (the provision of wildlife, viewing areas etc.) provide indirect cultural and social benefits. Most RSPB reserves are designed to provide such benefits alongside, and associated with, biodiversity. On the 22 focal reserves, 18 offered facilities including trails, car parks, media publicity, talks in the community, reserve viewpoints or guided walks, with 21 offering opportunities to volunteer to assist with work on the reserve. Educational facilities, visitor/information centres, cafes, public footpaths, toilets or bird-watching hides were provided on 13 of the reserves. The small numbers of sites that are not capable of supporting visitors do not provide facilities or record the number of visitors received, but 18 of the 22 reserves supported a total of 483,716 visits in 2007–2008.

Identifying how people are benefiting from the management activities and facilities is critical for linking conservation with wider societal welfare. On the six reserves where questionnaire surveys have been carried out, 47% of respondents selected birds as their main motivation for visiting the reserve, 44% selected wildlife and nature as their main motivation, while walking and other outdoor activities was the main motivation for 7.5%. The remaining 1.5% had either no specific interest in the reserve or did not answer this question (Fig. 4). Thus, the presence of wildlife on the reserves is a key source of attraction for the great majority of visitors. The distances travelled by visitors to reserves also varied, with over two thirds of the visits made locally or on day trips from home, ~20% from holiday makers near to the reserve and only 5% from holiday makers on day trips from more distant locations.

The types of visits and distances travelled by visitors varied among different types of reserves (Fig. 4). On the three coastal reserves that are more remote from urban centres (Minsmere, Dungeness and Titchwell), the recreational services provided by RSPB

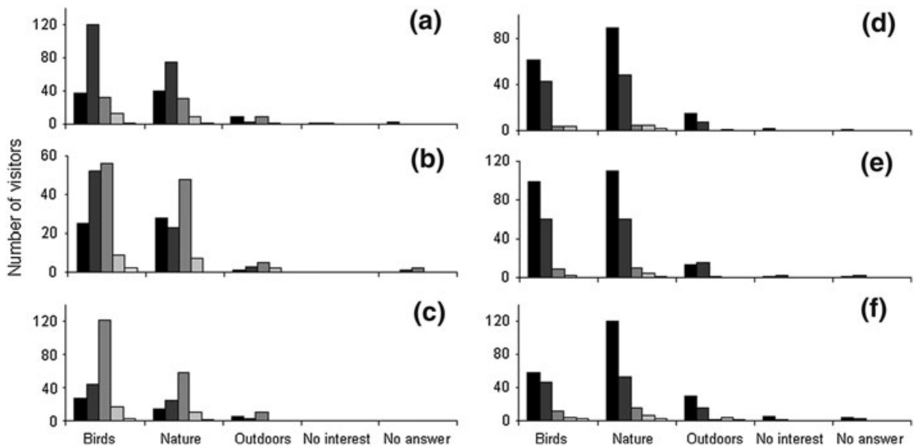

Fig. 4 Variation in numbers of visitors from different locations (local = *black*, day trip = *dark grey*, holidaying locally = *mid grey*, holidaying elsewhere = *pale grey*, no answer = *white*) stating different motivations for visiting three coastal (**a** Dungeness, $x^2_{16} = 31.2$, $P < 0.02$; **b** Minsmere, $x^2_{16} = 18.4$, $P < 0.3$; **c** Titchwell, $x^2_{16} = 21.6$, $P < 0.25$) and three inland (**d** Rainham Marsh, $x^2_{16} = 30.8$, $P < 0.03$; **e** Rye Meads, $x^2_{16} = 28.3$, $P < 0.03$; **f** Pulborough Brooks, $x^2_{16} = 128.2$, $P < 0.001$) RSPB wetland reserves in lowland England (see text for details)

management activities were typically enjoyed by holiday-makers or day-trippers, whereas on inland sites closer to urban areas (Pulborough Brooks, Rainham Marsh and Rye Meads), a far greater proportion of visitors lived locally. Though this might not be surprising, visitors to the more remote, coastal reserves were also significantly more likely to identify birds as the main motivation for the visit compared to nature, the outdoors or no interest ($x^2 = 77.9$, df $= 3$, $P < 0.001$). In addition, across all sites, visitors travelling non-locally were significantly more likely to identify birds as their main motivation for the visit, whereas local visitors were motivated more by general nature and outdoors interests ($x^2 = 41.05$, df $= 3$, $P < 0.001$). The extensive use of these sites by local people as well as holiday visitors with a particular bird interest (Fig. 4) indicates the broad cultural attraction of these sites. The good being delivered by these sites (birds of particular interest) directly hinges upon the explicit management activities undertaken.

Discussion and conclusion

Interventions for biodiversity conservation in highly modified landscapes typically consist of (1) the protection of highly valued habitat and its associated species from development and other threats, (2) the intervention management of protected sites, aimed at maintaining the habitat and its characteristic species considered to be of particular value, and (3) providing opportunities for people to enjoy the species and habitats at the focus of conservation activities. In this case study, we have seen that the dedicated conservation reserves owned or managed by RSPB do deliver on these traditional objectives. The success of these species-led actions is indicated by increases in populations of target bird species while national populations have generally continued to decline, and large numbers of visitors being attracted to these reserves, with a high percentage of people visiting reserves motivated by seeing birds. The latter also suggests the potential for significant economic benefits to local communities derived from species-led conservation actions (although local economies may also receive dis-benefits via interventions, see above).

Here, we tried to move beyond an assessment of the traditional goals and get a better understanding of the consequences of species-led conservation actions for other ecosystem services (Fig. 1b, c). Despite the context-dependence of the relationship between management and ecosystem service delivery, potential impacts on other services of the management toolbox used on the lowland wetland sites can be identified. Livestock grazing has potentially large effects on water and greenhouse gas related services, but there is little scope to alter management without compromising species objectives. There is little potential to alter reedbed management, for instance by making reed cutting more commercial, without compromising conservation objectives, and this would have limited influence anyway due to the relatively small area over which such management is practiced. There is some potential to alter woodland and scrub management, but again this would likely have limited influence due to the relatively small area over which such management is practiced. Finally, management of water levels potentially has large effects on provision of several services and there does appear to be some scope to align this objective with conservation objectives.

Whilst there are clear trade-offs between maintaining high water levels to achieve biodiversity conservation objectives, and low water levels to maximise flood storage benefits and minimise emissions of some greenhouse gases, the greatest potential for enhancing the provision of other services associated with species-led conservation management appears to be the creation and management of grazed washlands (Rhymer et al. 2010).

Washlands have higher potential capacity for flood storage than un-embanked river floodplains, which become inundated during small flood events. This inundation will reduce their capacity to store water from larger-scale, and potentially more economically damaging flood events. By contrast, water can be prevented from entering washlands during small-scale flood events, so maximising their flood storage potential for larger-scale flooding, while maintaining biodiversity conservation interest at other times.

Potential conflicts between biodiversity and flood risk could be partially resolved by careful design of reserve landscapes (e.g. see Morris et al. 2004). Excavating deeper pools would allow water levels to be kept relatively low, while still providing wetland habitat for birds, invertebrates and plants. Including areas of higher ground within washlands will increase the likelihood of some areas remaining unflooded and suitable for breeding waders, grazing wildfowl, and non-aquatic wetland invertebrates, during periods of flooding. Additional areas of compensatory unflooded wet grassland could also be created adjacent to washlands and floodplains to provide at least some areas for birds to nest and feed on, when the rest of the washland or floodplain is flooded. While this potential exists, the actual financial costs of this type of management drive its feasibility. Also, there is a potential trade-off where deep pools might be better for phosphorous retention, but might not be as valuable for biodiversity (Hansson et al. 2005).

While the findings presented here are mainly qualitative (Table 3), the process itself generated several insights for identifying potential synergies and trade-offs between species-led conservation and other ecosystem service provision, as well as identifying steps to quantify these services. The process used here was (1) identifying conservation objectives, (2) identifying the type and magnitude of management interventions to achieve the conservation objectives and (3) using expertise to scope how and which ecosystem services are affected by such management. This last step provides readily usable information to reserve managers, such as an understanding of where and when greenhouse gas fluxes may be greatest.

The very action of identifying how management might affect ecosystem service provision also provides guidance to the key measurements that need to be made for a fuller quantitative understanding of the possible synergies and tradeoffs. For example, in this case study, grazing on wet grasslands was a key management tool for species delivery. From Table 3 we know that this can have at least three effects on greenhouse gas fluxes: (1) it can increase carbon storage through soil compaction, (2) increase N_2O emissions via compaction of water logged soils, and (3) increase methane output via enteric fermentation. These effects highlight three places where field measurements or model estimations are required in order to quantify the *net* change in greenhouse gas fluxes as a result of grazing management. Knowing the *net* change reveals whether, in wet grasslands, grazing management increases greenhouse gas emissions or decreases them, effectively telling us whether this management practice leads to "arrow b" or "arrow c" in Fig. 1 with regards to greenhouse gas fluxes. We can perform the same exercise for each management intervention and sum the biophysical results to gain understanding of how the whole management portfolio affects an individual service.

We suggest that the process of interrogating management activities through an ecosystem services framework can lead to a better understanding of the net social benefits a protected area produces. Generating information such as that shown in Table 3 can highlight not only the interactions of interest between species-led interventions and wider ecosystem service delivery, but also the processes that need to be measured to get a fuller quantitative understanding of these interactions. Since species-led conservation is typical in landscapes highly modified by human intervention, and many conservation organisations are now using ecosystem services as an additional rationale for conservation, we suggest that the process used here could become a useful scoping exercise for site managers. It has the potential for improving ecosystem service benefits, and minimising ecosystem service dis-benefits derived from species-led management interventions.

Acknowledgments This work was conducted during a U.K. Population Biology Network (UKPopNet) funded project (Ecosystem service delivery by real-world conservation approaches: a scoping study for a lowland wetlands). We thank Dave Raffaelli, Katie Bolt, Alistair Burn, Debbie Coldwell, Mark Crick, Rob Cunningham, Rob George, Jo Gilbert, R. Kerry Turner, Sally Mackenzie, Mark Smart, Gwyn Williams, and Simon Wotton for stimulating and helpful discussion and comments during this research project.

References

Andam KS, Ferraro PJ et al (2010) Protected areas reduced poverty in Costa Rica and Thailand. Proc Natl Acad Sci USA 107(22):9996–10001

Anderson BJ, Armsworth PR, Eigenbrod F, Thomas CD, Gillings S, Heinemeyer H, Roy DB, Gaston KJ (2009) Spatial covariance between biodiversity and other ecosystem service priorities. J Appl Ecol 46:888–896

Andrews JE, Burgess D et al (2006) Biogeochemical value of managed realignment, Humber estuary, UK. Sci Total Environ 371(1–3):19–30

Balmford A, Bruner A, Cooper P, Costanza R, Farber S, Green RE, Jenkins M, Jefferiss P, Jessamy V, Madden J, Munro K, Myers N, Naeem S, Paavola J, Rayment M, Rosendo S, Roughgarden J, Trumper K, Turner RK (2002) Economic reasons for conserving wild nature. Science 297:950–953

Bartlett KH, Harriss RC (1993) Review and assessment of methane emissions from wetlands. Chemosphere 26:261–320

Beintema AJ, Müskens GJDM (1987) Nesting success of birds breeding in Dutch grasslands. J Appl Ecol 24:743–758

Brix H, Schierup H-H, Arias CA (2007) Twenty years experience with constructed wetland systems in Denmark—what did we learn? Water Sci Technol 56(3):63–68

Bullock A, Acreman M (2003) The role of wetlands in the hydrological cycle. Hydrol Earth Syst Sci 7:358–389

Butchart SHM, Walpole M et al (2010) Global biodiversity: indicators of recent declines. Science 328(5982):1164–1168

Cook HF (2007) Floodplain nutrient and sediment dynamics on the Kent Stour. Water Environ J 21:173–181

Cooper C (2009) What can we learn from old wetlands? Lessons that have been learned and some that may have been forgotten over the past 20 years. Desalination 246:11–26

Daily GC, Polasky S, Goldstein J, Kareiva PK, Mooney HA, Pejchar L, Ricketts TH, Salzman J, Shallenberger R (2009) Ecosystem services in decision making: time to deliver. Front Ecol Environ 7:21–28

DEFRA (2007) Securing a healthy natural environment: an action plan for embedding an ecosystems approach. http://www.defra.gov.uk/wildlife-countryside/pdf/natural-environ/eco-actionplan.pdf

Dobbie KE, Smith KA (2003) Nitrous oxide emission factors for agricultural soils in Great Britain: the impact of soil water-filled pore space and other controlling variables. Glob Change Biol 9:204–218

Eglington SM, Gill JA et al (2009) Habitat management and patterns of predation of Northern Lapwings on wet grasslands: the influence of linear habitat structures at different spatial scales. Biol Conserv 142(2):314–324

Egoh B, Reyers B, Rouget M, Bode M, Richardson DM (2009) Spatial congruence between biodiversity and ecosystem services in South Africa. Biol Conserv 142:553–562

English Nature (2001) Sustainable flood defence: the case for washlands. No. 406 Research Report prepared for EN by Risk Policy Analysts Limited, Peterborough

English Nature (2010) Summary conditions for SSSI sites available at: http://www.english-nature.org.uk/special/sssi/reportAction.cfm?Report=sdrt15&Category=N&Reference=0

Ferguson C, de Roda Husman AM, Altavilla N, Deere D, Ashbolt N (2003) Fate and transport of surface water pathogens in watersheds. Crit Rev Environ Sci Technol 33(3):299–361

Gilbert G, Gibbons DW, Evans J (1998) Bird monitoring methods: a manual of techniques for key UK species. RSPB/British Trust for Ornithology, The Wildfowl and Wetlands Trust, Joint Nature Conservation Committee, Institute of Terrestrial Ecology and The Seabird Group

Gilbert G, Tyler G, Smith KW (2005) Behaviour, home range size and habitat use by male Great Bittern *Botaurus stellaris* in Britain. Ibis 147:533–543

Gilbert G, Tyler GA, Dunn CJ, Ratcliffe N, Smith KW (2007) The influence of habitat management on the breeding success of the Great Bittern *Botaurus stellaris* in Britain. Ibis 149:53–66

Green RE (1996) Factors affecting the population density of the corncrake *Crex crex* in Britain and Ireland. J Appl Ecol 33:237–248

Hansson LA, Bronmark C, Nilsson PA, Bjornsson KA (2005) Conflicting demands on wetland ecosystem services: nutrient retention, biodiversity or both. Freshw Biol 50:705–714

Harrison PA (2010) Ecosystem services and biodiversity conservation: an introduction to the RUBICODE project. Biodivers Conserv (*OnlineFirst*). doi:10.1007/s10531-010-9905-y

Hess TM, Holman IP, Rose SC, Rosolova Z, Parrott A (2010) Estimating the impact of rural land management changes on catchment runoff generation in England and Wales. Hydrol Proc. doi:10.1002/hyp.7598

Holman IP, Hollis JM, Bramley ME, Thompson TRE (2003) The contribution of soil structural degradation to catchment flooding: a preliminary investigation of the 2000 floods in England and Wales. Hydrol Earth Syst Sci 7(5):754–765

Houghton JT, Filho LGM, Callander BA, Harris N, Kattenburg A, Maskell K (1995) The science of climate change: contribution of working group I to the second assessment report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge, 584 pp

Jarvie HP, Withers PA, Hodgkinson R, Bates A, Neal M, Wickham HD, Harman SA, Armstrong L (2008) Influence of rural land use on streamwater nutrients and their ecological significance. J Hydrol 350:166–186

Millennium Assessment (2005) Millennium Ecosystem Assessment. Island Press, Washington, DC

Morris J, Hess TM, Gowing DJ, Leeds-Harrison PB, Bannister N, Wade M, Vivash RM (2004) Integrated washland management for flood defence and biodiversity. Report to Department for Environment, Food and Rural Affairs & English Nature. Cranfield University at Silsoe, Bedfordshire, UK, March 2004

Morris J, Bailey AP, Lawson CS, Leeds-Harrison PB, Alsop D, Vivash R (2008) The economic dimensions of integrating flood management and agri-environment through washland creation: a case study from Somerset, England. J Environ Manag 88:373–381

Naidoo R, Balmford A, Costanza R, Fisher B, Green RE, Lehner B, Malcolm TR, Ricketts TH (2008) Global mapping of ecosystem services and conservation priorities. Proc Natl Acad Sci USA 105:9495–9500

Nelson E, Mendoza G, Regetz J, Polasky S, Tallis H, Cameron DR, Chan KMA, Daily GC, Goldstein J, Kareiva PM, Lonsdorf E, Naidoo R, Ricketts TH, Shaw MR (2009) Modeling multiple ecosystem services, biodiversity conservation, commodity production, and tradeoffs at landscape scales. Front Ecol Environ 7:4–11

O'Brien M, Smith KW (1992) Changes in the status of waders breeding on wet lowland grasslands in England and Wales between 1982 and 1989. Bird Study 39:165–176

O'Connell PE, Beven KJ, Carney JN, Clements RO, Ewen J, Fowler H, Harris GL, Hollis J, O'Donnell GM, Packman JC, Parkin A, Quinn PF, Rose SC, Shepherd M, Tellier S (2004) Review of impacts of rural land management on flood generation. Impact Study Report. Defra R&D Technical Report FD2114/TR. Defra, London

Peacock CE, Hess TM (2004) Estimating evapotranspiration from a reed bed using the Bowen ratio energy balance method. Hydrol Process 18:247–260

Posthumus H, Rouquette JR, Morris J, Gowing DJG, Hess TM (2010) A framework for the assessment of ecosystem goods and services; a case study on lowland floodplains in England. Ecol Econ 69:1510–1523

Ratcliffe N, Schmitt S, Whiffin M (2005) Sink or swim? Viability of a black-tailed godwit population in relation to flooding. J Appl Ecol 42:834–843

Redford KH, Adams WM (2009) Payments for ecosystem services and the challenge of saving nature. Conserv Biol 23:785–787

Rey Benayas JM, Newton AC, Diaz A, Bullock JM (2009) Enhancement of biodiversity and ecosystem services by ecological restoration: a meta-analysis. Science 325:1121–1124

Rhymer CM, Robinson RA, Smart J, Whittingham MJ (2010) Can ecosystem services be integrated with conservation? A case study of breeding waders on grassland. Ibis 152:698–712

Rodrigues ASL (2006) Are global conservation efforts successful? Science 313:1051–1052

RSPB (2009) Naturally, at your service: Why it pays to invest in nature. RSPB, Sandy

Smart J, Gill JA et al (2006) Grassland-breeding waders: identifying key habitat requirements for management. J Appl Ecol 43(3):454–463

Smart J, Amar A et al (2008) Changing land management of lowland wet grasslands of the UK: impacts on snipe abundance and habitat quality. Anim Conserv 11(4):339–351

Smith KA, Ball T, Conen F, Dobbie KE, Massheder J, Rey A (2003) Exchange of greenhouse gases between soils and atmosphere: interactions of soil physical factors and biological processes. Eur J Soil Sci 54:779–791

Tyler GA, Smith KW, Burgess DJ (1998) Reedbed management and breeding Bitterns *Botaurus stellaris* in the UK. Biol Conserv 86:257–266

Turner RK, Daily GC (2008) The ecosystem services framework and natural capital conservation. Environ Resour Econ 39:25–35

Turner RK, Paavola J, Cooper P, Farber S, Jessamy V, Georgiou S (2003) Valuing nature: lessons learned and future research directions. Ecol Econ 46:493–510

Turner RK, van den Bergh JCJM, Soderqvist T, Barendregt A, van der Straaten J, Maltby E, van Ierland EC (2000) Ecological-economic analysis of wetlands: scientific integration for management and policy. Ecol Econ 35:7–23

Van der Vliet RE, Schuller E, Wassen J (2008) Avian predators in a meadow landscape: consequences of their occurrence for breeding open-area birds. J Avian Biol 39:523–529

Vymazal J (2007) Removal of nutrients in various types of constructed wetlands. Sci Tot Environ 380:48–65

Whiting GJ, Chanton JP (2001) Greenhouse carbon balance of wetlands: methane emission versus carbon sequestration. Tellus 53B:521–528

Wilcove DS (2008) No way home: the decline of the world's great animal migrations. Island Press, Washington, DC

Wilcove DS, Rothstein D et al (1998) Quantifying threats to imperiled species in the United States. Bioscience 48(8):607–615

Wilson AM, Vickery JA, Brown A, Langston RHW, Smallshire D, Wotton S, Vanhinsbergh D (2005) Changes in the numbers of breeding waders on lowland wet grasslands in England and Wales between 1982 and 2000. Bird Study 52:55–69

Wilson A, Vickery J, Pendlebury C (2007) Agri-environment schemes as a tool for reversing declining populations of grassland waders: mixed benefits from Environmentally Sensitive Areas in England. Biol Conserv 136:128–135

Withers PJA, Sharpley AN (2008) Characterization and apportionment of nutrient and sediment sources in catchments. J Hydrol 350:127–130

Wotton S, Brown A, Burn A, Cunningham R, Dodd A, Droy N, Gilbert G, Rees S, White G, Gregory R (2009) Boom or bust—a sustainable future for reedbeds and Bitterns? Br Wildl 20:305–315