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A review and application of the evidence for nitrogen impacts on ecosystem services

L. Jones[a,*], A. Provins[b], M. Holland[c], G. Mills[a], F. Hayes[a], B. Emmett[a], J. Hall[a], L. Sheppard[d], R. Smith[d], M. Sutton[d], K. Hicks[e], M. Ashmore[e], R. Haines-Young[f], L. Harper-Simmonds[b]

[a] Centre for Ecology and Hydrology (CEH), Environment Centre Wales, Deiniol Road, Bangor, LL57 2UW, UK
[b] eftec, Economics for the Environment Consultancy, 73–75 Mortimer Street, London, UK
[c] EMRC, 2 New Buildings, Whitchurch Hill, Reading, RG8 7PW, UK
[d] Centre for Ecology and Hydrology (CEH), Bush Estate, Edinburgh, UK
[e] Stockholm Environment Institute at York, Grimston House, University of York, Heslington, York, YO10 5DD, UK
[f] Centre for Environmental Management, School of Geography, University of Nottingham, Nottingham, NG7 2RD, UK

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ABSTRACT

Levels of reactive nitrogen (N) in the atmosphere have declined by around 25% in Europe since 1990. Ecosystem services provide a framework for valuing N impacts on the environment, and this study provides a synthesis of evidence for atmospheric N deposition effects on ecosystem services. We estimate the marginal economic value of the decline in N deposition on six ecosystem services in the UK. This decline resulted in a net benefit (Equivalent Annual Value) of £65 m (£5 m to £123 m, 95% CI). There was a cost (loss of value) for provisioning services: timber and livestock production of −£6.2 m (−£3.5 m to −£9.2 m, 95% CI). There was a cost for CO_2 sequestration and a benefit for N_2O emissions which combined amounted to a cost for greenhouse gas regulation of −£15.7 m (−£4.5 m to −£30.6 m). However, there were benefits for the cultural services of recreational fishing and appreciation of biodiversity, which amounted to £87.7 m (£13.1 m to £163.0 m), outweighing costs to provisioning and regulating services. Knowledge gaps in both the under-pinning science and in the value-transfer evidence prevent economic valuation of many services, particularly for cultural services, providing only a partial picture of N impacts which may underestimate the benefits of reducing N deposition.

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* Corresponding author. Tel.: +44 1248 374500.
E-mail address: lj@ceh.ac.uk (L. Jones).

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1. Introduction

Levels of reactive nitrogen (N) in the atmosphere have increased globally since the 1940s as a result of man's activities (Galloway et al., 2008). The main sources of oxidised N compounds are vehicle emissions, industry and domestic combustion, while reduced N compounds, primarily ammonia, derive from agriculture sources such as manure and fertiliser volatilisation. Nitrogen is a basic nutrient required for growth, and most semi-natural systems are N-limited (Vitousek et al., 1997). Increased N deposition in the last 70 years has caused widespread adverse impacts on biogeochemical cycling and biodiversity in semi-natural systems as a result of both eutrophication and acidification, which have been well studied (e.g. Duprè et al., 2010; Phoenix et al., 2012; Sutton et al., 2011a). However, since N stimulates plant growth, deposition of this nutrient may be seen as beneficial for human production systems, e.g. by increasing forest growth (De Vries et al., 2009).

Across Europe, emissions of N have now declined by 25% since around 1990 due to policy measures to reduce industrial and vehicular emissions of oxidised N, and to reduce ammonia emissions from agriculture (Oenema et al., 2011). However, the effect of this decline in emissions has not been systematically evaluated across a wide range of sectors.

Ecosystem Services frameworks are emerging as a way of capturing the wider effects of policy decisions or evaluating land use change in order to more comprehensively take into account the range of impacts on the environment, and on the benefits humans receive from it (Turner and Daily, 2008). However, although rapidly developing, much of the conceptualisation around ecosystem services, and the data required to quantify them, do not readily marry to existing experimental data, and the links to ecological processes are poorly defined. Applications to new situations are often largely qualitative, based on expert judgement or assumptions, and lack supporting evidence from the literature. When examined in more detail, the literature reveals far greater complexity to what are presented as simple relationships. There is a need to bridge this gap in scientific understanding between ecosystem processes and ecosystem service delivery.

Valuation of ecosystem services, monetary and non-monetary, increasingly feeds into decision making processes (Fisher et al., 2009). Assigning an economic value to N pollution impacts has been conducted in some studies e.g. valuation of ammonia (NH_3) impacts on human health (Holland et al., 2005; Watkiss, 2008), and for impacts of N from agriculture (Sutton et al., 2011b). Detailed cost-benefit approaches have been applied in The European Nitrogen Assessment for the impacts of nitrogen in Europe (Brink et al., 2011), and for the effects of water quality legislation in Chesapeake Bay, USA (Morgan and Owens, 2006). An ecosystem services framework has been proposed for ammonia pollution (Smart et al., 2011), but has not yet been applied in detail. A key challenge of applying an ecosystem services framework to the valuation of air pollution impacts is that it requires a full understanding and quantification of the impact pathway: from changes in emissions, to deposition and its consequent effects on ecosystem processes and how those changes affect ecosystem service provision and the goods and benefits arising from them.

Therefore, in this paper we aim to review the published evidence supporting N impacts on ecosystem services and show how improved understanding of those links can be used to guide valuation of impacts. Firstly, we describe the main mechanisms of N impacts on ecological systems, and then make explicit the conceptual links between N and supporting, provisioning, regulating and cultural ecosystem services. The study then conducts a marginal cost analysis using examples in a UK context, comparing the impact of a net reduction in N deposition from 1987 to 2005, using 1987 as the reference year. For this we use the typology of final ecosystem services (*sensu* Fisher et al., 2008) as developed in the UK National Ecosystem Assessment (UK NEA, 2012).

2. Mechanisms of N impacts on ecological processes

Nitrogen impacts are manifested through three principal mechanisms: eutrophication, acidification and direct toxicity (Bobbink et al., 2010). We briefly describe these mechanisms here, and show the conceptual links to ecosystem services for each mechanism (Figs. 1–3), based on the wide range of impacts on

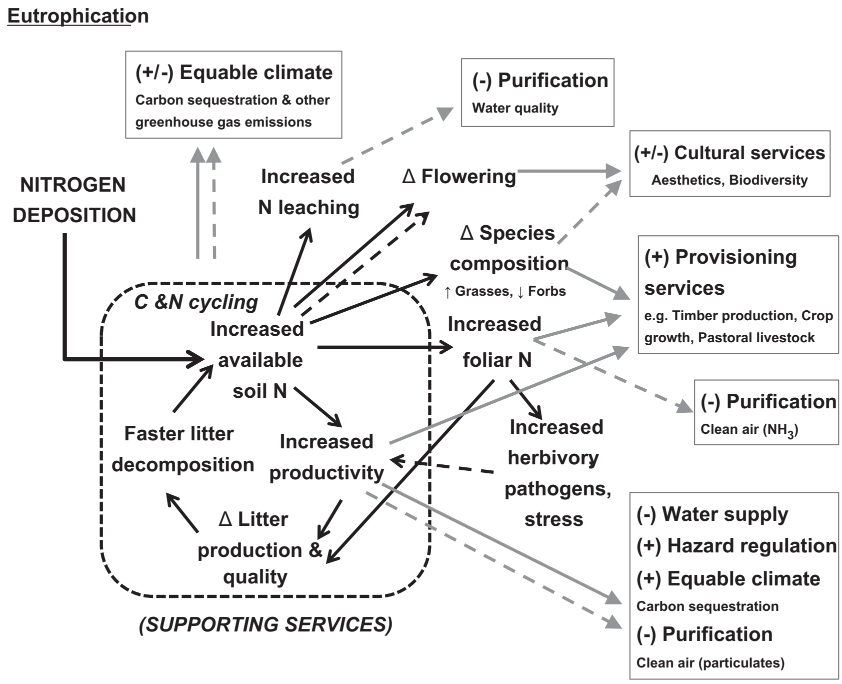

Fig. 1. Process-based impact pathway for *eutrophication*. Black arrows indicate process-based links, grey arrows show links to ecosystem services, where + and − indicate the nature of relationship and examples are given in small type. Solid arrows represent positive relationships and dashed arrows negative relationships. The dotted line box encompasses processes linked to C and N cycling (=Supporting Services). Impacts on species composition are generalised to increases in graminoids and decreases in forbs, but in reality are much more complex.

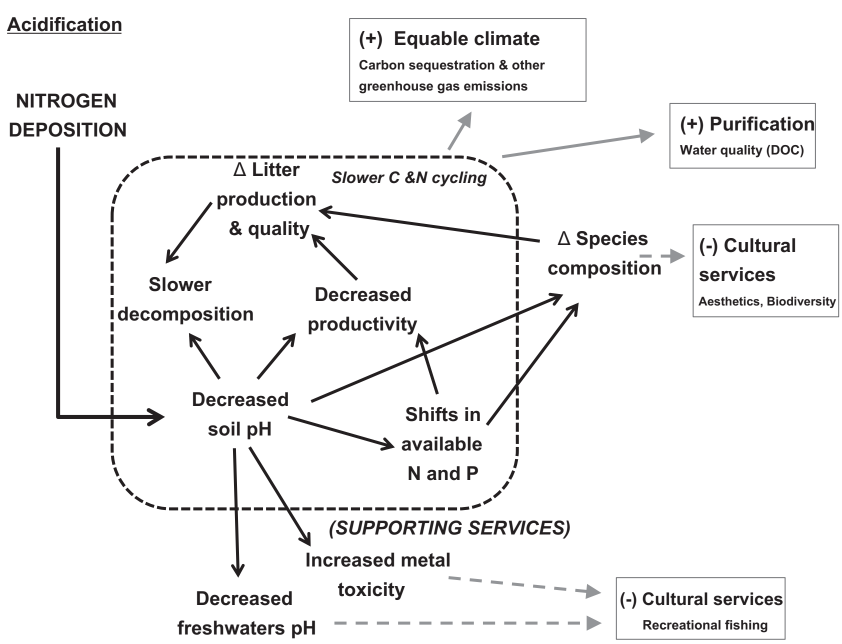

Fig. 2. Process-based impact pathway for *acidification*. For legend, see Fig. 1.

ecological processes. The evidence for these impacts is discussed in more detail in Sections 3 and 4.

Eutrophication (Fig. 1) of oligotrophic (i.e. nutrient poor) habitats occurs where there is excess nutrient availability, above the natural, pre-industrial levels. Since N is a nutrient, it increases the quantity of available N in the soil, stimulating plant productivity and rates of nutrient cycling in N limited terrestrial and aquatic systems (Vitousek et al., 1997; Fisher, 2003). These processes governing the biogeochemical cycling of carbon (C) and N are analogous to the supporting services, and are represented within the dashed line box of Fig. 1. Changes in primary productivity and accumulation of N in soils (Jones et al., 2008) subsequently affect other soil or water mediated processes such as N leaching, or biological processes including flowering, alteration of competitive relationships between species, nutrient imbalances or nitrogen saturation, and indirect impacts mediated by changes in stoichiometry (Clark and Tilman, 2008; Sala et al., 2000). These in turn affect a range of provisioning, regulating and cultural ecosystem services, shown in Fig. 1.

Nitrogen contributes to acidification of soils and freshwater systems (Fig. 2). Historically this acidification was primarily due to

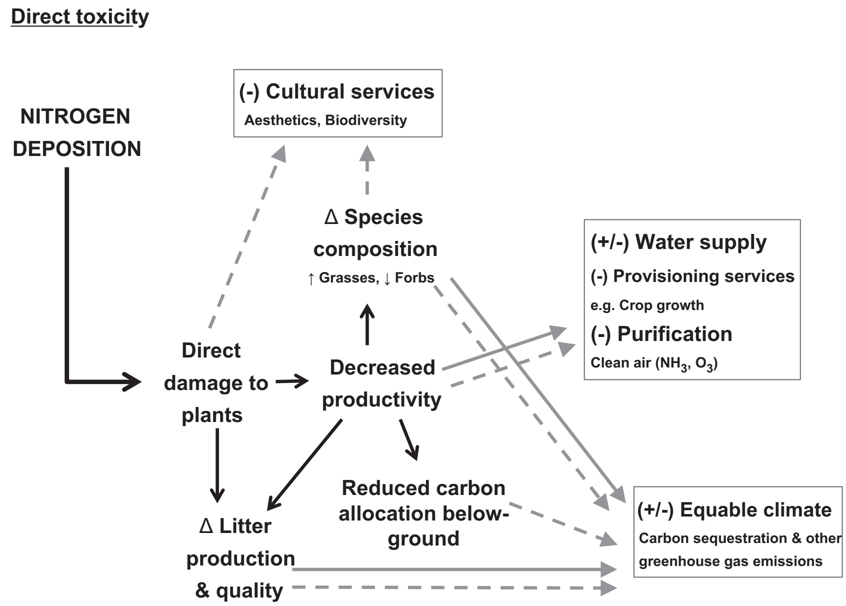

Fig. 3. Process-based impact pathway for *direct toxicity* (incorporating NO_x and NH_3 effects). For legend, see Fig. 1.

high sulphur deposition. However, since sulphur deposition has declined dramatically across Europe, N now makes a greater contribution to acidity than sulphur at current deposition levels (ROTAP, 2012). Uptake and assimilation of ammonium by plant roots and the process of nitrification, and subsequent leaching of nitrate cause acidification of the soil (Gundersen and Rasmussen, 1990). Acidification impacts occur through toxicity effects on aquatic and terrestrial organisms due to exceedance of biological and chemical thresholds of soil pH, and increased mobilisation of toxic ions such as Al^{3+}. Effects on plant growth also occur through soil pH controls on P availability (Kooijman et al., 1998; Brady and Weil, 1999), which indirectly alters plant productivity. Impacts on regulating services occur directly through lowered soil pH and slower rates of biogeochemical cycling and organic matter decomposition, while impacts on cultural services are mediated through changes in abundance or diversity of organisms such as fish or changes in plant growth and community composition.

Direct toxicity (Fig. 3) is caused by the gaseous forms of N as ammonia or as nitrogen dioxide. At very high concentrations, nitrogen dioxide is toxic to plant growth, but in many cases the toxicity is due to chronic exposure, i.e. annual dose, rather than acute toxicity. In much of Europe, concentrations of nitrogen dioxide are below the critical levels defined in LRTAP Convention (2010), with the exception of some urban areas or close to major roads and large point sources. Ammonia is also toxic to plant growth at high concentrations. The critical level for ammonia for vegetation is an annual mean of 3 $\mu g\, m^{-3}$ to protect semi-natural vegetation, and 1 $\mu g\, m^{-3}$ to protect sensitive lichens and bryophytes (Cape et al., 2009). The majority of toxicity impacts are mediated by reduced plant growth with negative impacts on provisioning services, with some indirect effects on species composition and on biogeochemical cycling (Fig. 3), which impact on regulating services and cultural services. Note that reduced plant growth may have positive impacts on some services such as water supply, and altered nutrient cycling may have positive or negative effects on greenhouse gas emissions. These are discussed in more detail in Section 4.

Each of these three mechanisms has an impact on a range of ecosystem services, for which we summarise the evidence below. There is an extensive literature spanning many decades on N impacts on ecosystem processes, and more recently on biodiversity. We do not aim to reproduce this, rather to summarise the main impacts that are of relevance to ecosystem service provision, with selected supporting references. We discuss first the impacts on supporting services, and then the implications for intermediate and final provisioning, regulating and cultural services.

3. Effects of N on supporting services

3.1. Primary production

Since N is the principal limiting nutrient (Vitousek et al., 1997), increases in N deposition usually increase net primary production (NPP) in terrestrial systems. This increased plant productivity fixes additional carbon through photosynthesis, which enters soil carbon pools through litter fall, root turnover and root exudates, contributing to nutrient cycling and ultimately soil formation. There are important exceptions, where climate or limitation of other nutrients, such as phosphorus, prevent a growth response to N, and where nutrient imbalances or direct toxicity reduce plant growth. In such cases N does not increase NPP, but may still accumulate in the system or have other impacts. In aquatic systems there is increasing evidence that freshwaters can be N limited or N and P co-limited, rather than P limited (Maberly et al., 2002). In N-limited aquatic systems, atmospheric N can increase primary productivity of plankton and macrophytes and decrease macrophyte diversity (e.g. Fenn et al., 2003; James et al., 2005).

3.2. Nutrient cycling

Nutrient cycling in terrestrial systems is generally understood to mean the rate of mineralisation of organic matter. Inorganic N from atmospheric deposition affects decomposition rates through litter quality *via* increased tissue N, i.e. reduced C:N ratio, and altered lignin concentrations, but responses are complex. In a meta-analysis, decomposition was stimulated by N where litter N or lignin content was low, but retarded where litter N or lignin contents were high (Knorr et al., 2005). In general, N deposition increases mineralisation rates, leading to faster turnover of N in the soil and greater availability of labile inorganic N for plant growth, further increasing primary production. Excess N can lead to nitrate (NO_3^-) leaching, if N availability exceeds biological demand (Emmett, 2007). By contrast, acidification below pH 4.5 can slow microbial decomposition, due to changes in bacterial community composition, and a shift from bacterial to increasing fungal dominance of the microbial decomposer community (Rousk et al., 2010). Increases in the solubility of toxic aluminium forms below pH 4.5 can also slow microbial activity and therefore nutrient cycling.

3.3. Soil formation

Soil formation occurs through accumulation of soil organic matter and through mineral weathering of rock. Organic matter accumulation depends on the balance of carbon inputs from plant litter (i.e. Net Primary Productivity, NPP) and the rate at which the plant matter is decomposed (i.e. nutrient cycling), both discussed above. Air pollution can alter both sides of the equation. The balance between production and decomposition is difficult to quantify and varies spatially due to temperature and soil moisture controls on decomposition rates and plant production, as well as land-use in many habitats. The net effects of N on carbon sequestration in soils are reasonably well understood for a range of habitats. In forests and heathlands N deposition generally increases soil organic matter accumulation (De Vries et al., 2009), in grasslands moderate fertilisation increases soil C stocks while intensive fertilisation decreases them (Soussana et al., 2004), while excess N deposition decreases soil C stocks in bogs, peatlands (Bragazza et al., 2006) and Alaskan tundra soils (Mack et al., 2004). Soil formation has direct implications for soil C sequestration, but underlies a range of other provisioning, regulating and to a lesser extent cultural services reliant on soil.

4. Effects of N on provisioning services

4.1. Food production in highly managed agricultural habitats – crops, livestock, dairy

Nitrogen stimulation of plant production can have positive effects for provisioning services linked directly to plant growth such as crops and feedstock production, including hay and silage, and indirectly via increased or faster weight gain in livestock (Jensen et al., 2011; Brink et al., 2011). In fertilised agricultural systems, including dairy and beef production on re-sown highly fertilised leys the relatively small amounts of atmospheric N input (deposition range in Europe ca. 5–40 kg N ha^{-1} yr^{-1} for non-forest systems) compared with typical agricultural additions of 10–200 kg N ha^{-1} yr^{-1} (Velthof et al., 2009) still represent a cost-saving in fertiliser input for farmers, who otherwise would have to add greater quantities of fertiliser. An analogous situation exists for sulphur following major declines in sulphur deposition in the UK, where sulphate additions are now required for some crops, especially brassicas (Zhao et al., 2003). Acidification and ammonia toxicity impacts are less likely in these fertile systems where plants have the capacity to upregulate assimilation, and soil pH is managed by farmers, while direct toxicity from nitrogen dioxide is unlikely at current concentrations, although may cause localised damage near point sources.

4.2. Food production – livestock in semi-natural grasslands

Improved (i.e. drained and fertilised) pastures and short-rotation leys are considered within fertilised agricultural systems above. Unimproved and semi-improved grasslands produce hay and provide lower-quality grazing land for extensive livestock production. Surprisingly, there is limited experimental evidence for increased grassland productivity due to elevated N deposition in semi-natural grasslands: at high N deposition, biomass production increased in acid grasslands, but showed no change or marginal increases in calcareous grasslands (Phoenix et al., 2003) possibly due to P limitation. Species compositional change driven by elevated N deposition may improve forage quality, and hence livestock production, by favouring nitrophilous graminoids (Maskell et al., 2009) and increases in tissue N content of some species (Plassmann et al., 2009). By contrast, acidification may reduce productivity in acidic grasslands, and may drive species compositional change (Stevens et al., 2010), reducing forage quality.

4.3. Food production – game in moorlands

Enhanced productivity, shoot growth and foliar N content of *Calluna vulgaris* with increased N have been shown in heathland habitats (Britton et al., 2008; Power et al., 2006). In upland habitats where *Calluna* is dominant this could have the potential to support larger populations of grouse or deer for shooting. However, an indirect consequence of N addition is acceleration of the *Calluna vulgaris* growth cycle, with more frequent management needed to prevent degradation (Terry et al., 2004). This is classed as a provisioning service here, but could alternatively be considered a cultural service with shooting as a leisure activity. Further acidification of these acidic habitats is unlikely to affect *Calluna vulgaris* growth, however ammonia toxicity may do so (Sheppard et al., 2008).

4.4. Wild food production – fungi, all habitats

The evidence surrounding N deposition effects on fungi and production of their fruiting bodies is complex. A review of the evidence suggests that N deposition can cause widespread reductions in certain fungal fruiting bodies, but effects on fungal abundance or diversity below ground are less clear (Wallenda and Kottke, 1998).

4.5. Fibre production – wool in semi-natural grasslands

The conceptual links and the reasoning for increased meat production in semi-natural grasslands, discussed above, may be transferred also to wool production. However, there is no evidence linking N deposition to changes in wool production or quality to our knowledge.

4.6. Fibre production – timber in woodlands

There is considerable interest in the impacts of N deposition on woodland productivity, centred around the issue of C sequestration in woodland, extensively reviewed in De Vries et al., (2008, 2009) and Sutton et al., (2008). Nitrogen can stimulate tree growth, and positive impacts of N as a nutrient outweigh negative impacts mediated by acidification (Solberg et al., 2009).

4.7. Genetic diversity of wild species

There is some evidence for effects of eutrophication, acidification and direct toxicity on some species groups. There are well documented negative impacts on wild (plant) species diversity, with further implications for invertebrates and other ecosystem components which depend on them (e.g. Bobbink et al., 2010). These are discussed further under cultural services below. Nitrogen deposition may reduce the ecological plasticity of some plant species, with potential implications for future genetic diversity under global change (Vergeer et al., 2008). There is some limited evidence for negative effects of eutrophication due to N on carabids and butterflies (Nijssen et al., 2001; WallisDeVries and

Van Swaay, 2006) and on soil microbial and fungal diversity (Lilleskov et al., 2002; Pardo et al., 2010).

4.8. Water supply

There is a conceptual link between enhanced vegetation growth and increased water use and interception by plants, which could result in decreased water fluxes into reservoirs and groundwater, potentially affecting the quantity of water available for extraction as drinking water. The magnitude of this flux is not known, but may warrant further work. For example, it is suggested that ozone effects on tree physiology may alter water volume in rivers of forested catchments (McLaughlin et al., 2007).

5. Effects of N on regulating services

5.1. Equable climate – C stocks in vegetation and soils, and net C sequestration rates

In this review we make a distinction between C stocks in vegetation and soils and C sequestration. Sequestration rates (i.e. annual flows of C) are more easily monetised than C stocks.

In most habitats, only soil C stocks are considered a long-term store, although in forests a degree of long-term C storage may occur in above-ground biomass in long-lived tree species, or in timber products with high longevity. Nitrogen causes increases in tree growth (Wamelink et al., 2009a), also discussed under provisioning services for timber above; while impacts on soil carbon stocks are also discussed under soil formation above. There is experimental evidence that N can increase soil C stocks in heathlands, grasslands and forests (Power et al., 1995; Phoenix et al., 2003; Vanguelova et al., 2007), but at moderate to high levels, may have adverse impacts in some habitats, reducing soil C in bogs for example (Bragazza et al., 2006), while impacts in other habitats are unknown.

The extrapolation to changes in net C sequestration rate over time is harder to quantify. However, by applying a range of techniques including net ecosystem exchange measurements, pollution gradient and chronosequence studies, tracer studies and modelling approaches, the longer-term implications (i.e. over timescales of 30–40 years or more) of N deposition on net C sequestration over time can be derived for forests and moorlands (De Vries et al., 2009), and some grasslands (Rowe et al., 2006; Jones et al., 2008). In the long term, acidification can impact C stocks and sequestration rates. The reduction in decomposition rates of recalcitrant litter produced by calcifuge species is greater than the reduction in mineralisation rates and plant growth. Therefore, the net carbon gain in soil increases (Berg and McClaugherty, 2007).

5.2. Equable climate – other greenhouse gas emissions

Emissions of the greenhouse gas nitrous oxide (N_2O) are strongly linked to N inputs, but denitrification and subsequent release of N_2O is controlled by soil temperature and moisture conditions. It is assumed that ca. 1% of deposited N is re-released as N_2O–N (IPCC, 2006), but higher emission rates have been shown, depending on ecosystem type (Skiba et al., 1998; Pilegaard et al., 2006). Nitrogen deposition can suppress the oxidation of methane in the soils of neutral to calcareous grasslands, forests and arable systems, thereby increasing emissions of this potent greenhouse gas (Hutsch et al., 1993), although the changes in these habitats are likely to be small.

5.3. Purification – clean air (particulates and gaseous pollution)

Plants both directly trap particulates on leaf surfaces, which is a physical process, and take up gaseous pollutants through deposition onto leaf surfaces and uptake via stomata (Freer-Smith et al., 1997; Beckett et al., 1998). In principle, larger plants provide greater filtering of particulates (Nowak and Crane, 2000). However, the relatively limited growth stimulation of plants by N is unlikely to drastically affect this aspect of air quality.

5.4. Purification – clean air (ammonia concentrations)

Nitrogen status of plants has a direct influence on the absorption and uptake of gaseous ammonia. At a certain level of N status, there is a compensation point when plant stomata no longer take up ammonia, and may release it (Dragosits et al., 2008). Thus the N status of the vegetation will influence the proportion of ammonia deposited and the distance it travels from a point source before it is deposited on vegetation. However, these impacts are difficult to quantify at large scale.

5.5. Purification – clean water (nitrates and dissolved organic carbon – DOC)

Nitrate leaching into freshwaters is a consequence of overloading the regulating service of purification naturally provided by soils of all habitats and becomes a water quality issue, with associated treatment costs for water companies producing clean drinking water. Relationships have been derived between N deposition and N export *via* leaching, for forests (Dise and Wright, 1995; Gundersen et al., 1998), moorlands (Curtis and Simpson, 2007), with more limited evidence for calcareous grasslands (Phoenix et al., 2003), but relationships differ due to soil and vegetation-mediated differences in N saturation and N cycling. It is possible to model N leaching at catchment scales and above (e.g. Kronvang et al., 2009).

Dissolved organic carbon (DOC) causes the brown colouring in many upland streams and lakes and is a water quality issue. This can be regarded as a dis-benefit of the effects of N deposition on the environment, as it represents a cost for water companies to remove the colouring from affected drinking water sources in the uplands. DOC concentrations are inversely related to acidity. DOC concentrations in freshwaters are rising following declines in S pollution (Evans et al., 2006), but acidity due to N also affects DOC concentrations (Butterbach-Bahl et al., 2011).

5.6. Hazard regulation, reduced flooding (rivers and coastal)

There are conceptual links between increased N deposition and reduced flooding in rivers, via increased plant growth with associated greater water use reducing run-off to rivers, but also increased macrophyte growth within lowland rivers reducing water flow rates. However, there are complex feedbacks associated with soil drying and unpredictable impacts on run-off. The links with coastal flooding have a stronger conceptual basis. Nitrogen deposition increases vegetation growth in mobile and semi-fixed coastal dunes (Jones et al., 2004), and in saltmarsh (Van Wijnen and Bakker, 1999). Taller and more vigorous vegetation in dunes binds the sand and is one factor promoting higher dune building, improving the flood defence capability of the leading dune (Barbier et al., 2008). In saltmarsh, vegetation acts to attenuate wave energy (Möller et al., 1999), and stimulation of vegetation growth by N deposition can potentially increase this service.

6. Effects of N on cultural services

6.1. Water-based leisure activities, and recreational fishing

In N limited lakes and freshwaters, there is the potential for N deposition to cause high algal growth adversely affecting amenity value through visual impacts, restriction of amenity uses such as boating and swimming e.g. due to toxic algal blooms (van der Molen et al., 1998; Camargo and Alonso, 2006). Eutrophication of water courses may also affect recreational fishing through alteration of aquatic food-webs and ultimately fish populations (Smith and Schindler, 2009). There is increasing understanding of the role of N limitation rather than P limitation in freshwater systems (Maberly et al., 2002; James et al., 2005), but the specific contribution of atmospherically deposited N to eutrophication is not easy to separate.

Acidification of freshwaters, which is in part due to N deposition, causes loss or damage to fish populations. Impacts of high concentrations of labile aluminium and hydrogen ions on fish reproductive capability include effects on egg production, hatching success and on physiological parameters such as osmo-regulation and gill function (Donaghy and Verspoor, 1997; Kroglund et al., 2008). Studies have shown links between acid neutralisation capacity (ANC) in waters and health of fish populations in Norway, with a critical threshold of 20 μeq L^{-1} ANC, based largely on impacts on Atlantic salmon in rivers and brown trout in lakes (Lien et al., 1996; Lydersen et al., 2004; Kroglund et al., 2008).

6.2. *Aesthetic appreciation of natural environment (flowering)*

Flowering of wild species is an important component of the attraction of wild habitats to the public, for example bluebell woods and wildflower meadows. Nitrogen deposition affects both the abundance and composition of flowering species, but also their individual flowering rates. Nitrogen manipulation experiments show that flowering of *Calluna vulgaris* in heathlands is enhanced by N (Power et al., 1995), while flowering in grasslands is reduced by N (O'Sullivan, 2008). Thus, responses to N are habitat specific. Response functions linking flowering to N can be derived, but their use in assessment of cultural importance depends on the valuation techniques available.

6.3. *Aesthetic appreciation of natural environment (weediness and visual damage)*

High N deposition around ammonia point sources, such as pig or chicken farms, in woodland areas dramatically alters the groundflora to that of nitrophilic species such as nettles *Urtica dioica* and the grass Yorkshire fog *Holcus lanatus* (Pitcairn et al., 2002). These species are common in disturbed ground, giving a 'weedy' appearance, in contrast to unimpacted more natural areas. It is possible that this weedy appearance is of less value to the public. The direct toxicity effects of ammonia on cultural services are probably limited to visual damage on plants close to point sources. High ammonia concentrations cause greying of *Calluna vulgaris* shoots, and increase sensitivity to stresses such as drought or winter desiccation which causes further die-back (Sheppard et al., 2008). However, to our knowledge the air pollution impacts on the aesthetic quality of the natural environment have not been subject to empirical research, and the magnitude of any potential effects have not been quantified.

6.4. *Appreciation of biodiversity*

In a wide range of terrestrial habitats, declines in plant species richness have been noted with N deposition globally (Bobbink et al., 2010; Pardo et al., 2010). These include acid grasslands (Stevens et al., 2004; Duprè et al., 2010), sand dune grasslands (Jones et al., 2004) and mixed grassland, heath and bog, and deciduous woodland (Maskell et al., 2009). However, in some habitats, loss of plant species diversity due to N has not been observed: calcareous grasslands (Van Den Berg et al., 2011), woodland epiphytes (Mitchell et al., 2005), Scottish montane habitats (RoTAP 2012) or in Racomitrium heath (Armitage et al., 2012). Although, in some of these habitats, other changes have occurred which damage their conservation status, such as shifts in calcareous grasslands towards domination by the grass *Brachypodium pinnatum* (Bobbink and Willems, 1987), and direct toxicity damage to the moss *Racomitrium lanuginosum* in Racomitrium heath (Jones et al., 2002; Pearce and van der Wal, 2002). This direct reduction in biodiversity in some habitats is likely to have adverse impacts on cultural services associated with appreciation of the natural environment.

There is increasing evidence that some of the negative effects on species richness and community composition previously ascribed to eutrophication may, in acid-sensitive habitats, be partially due to acidification of soils, by both N and S. This has been shown in national surveys of acid grasslands (Maskell et al., 2009; Stevens et al., 2010) and heathland (Maskell et al., 2009). Species loss due to N is not confined solely to vegetation, for example the decline of more charismatic taxa such as red shrike *Lanius collaris* in the Netherlands and Denmark linked to changes in prey items (Nijssen et al., 2001), but is harder to quantify or value without specific studies.

A number of studies show shifts in plant community composition or species biomass along ammonia concentration gradients away from intensive livestock units, for woodland groundflora (e.g. Pitcairn et al., 2002) and sand dune grassland (Jones et al., 2013). Direct evidence for loss of species richness is provided by an ammonia release experiment on a bog system where *Sphagnum capillifolium*, *Cladonia* spp. and *Calluna vulgaris* all disappeared at the higher ammonia concentrations (Sheppard et al., 2009).

In freshwaters, reduced macrophyte species richness has been associated with winter nitrate concentrations in lakes (van der Molen et al., 1998). Freshwater acidification, in part due to N deposition, has caused changes in epilithic diatoms, aquatic macrophytes, macroinvertebrate communities as well as fish and bird populations (Ormerod and Durance, 2009). These predominantly adverse impacts on biodiversity are likely to have had a negative impact on the cultural services associated with human use and appreciation of rivers. In contrast to terrestrial systems, acidification in freshwaters has a marked impact on charismatic species such as salmon, trout and the birds that feed in rivers. Such effects may be more noticeable by the general public than most eutrophication effects in terrestrial systems.

Diversity loss is likely to have higher resonance with the public than shifts in community composition, although both are important indicators of N damage to ecosystems. Shifts in species dominance are much more common than loss of species richness and may drive previously widespread species to increasing rarity with the potential for future species loss. Preferences for conservation of species can be related to rarity, beauty or other cultural factors. These are all motivations that typically relate to both non-use and use values linked to appreciation of the natural environment and diversity.

7. Methods

In this section, we show how an improved understanding of the links between atmospheric N deposition and ecosystem services can be used to guide valuation of those impacts. Using historical changes in deposition in the UK over a twenty year period in an ex-post assessment, we explore the impact of declines in N deposition on the value of six ecosystem services: two provisioning (timber and grassland production), two regulating (carbon sequestration and N_2O emissions) and two cultural services (recreational fishing and appreciation of biodiversity).

7.1. Overview

Marginal change in the value of ecosystem services was calculated based on a scenario comparison of impacts resulting from the change in N deposition over the period 1987–2005 (Jones et al., 2012). The scenario compared observed changes in pollutant emissions from 1987 to 2005, versus maintenance of 1987-levels of emissions as the counterfactual. In other words, what had been the benefit of observed reductions in pollutant emissions during the period 1987–2005. This interval covers the main period for which accurate UK emissions and deposition data have been calculated using a consistent methodology. Over this period N emissions and N deposition increased slightly until 1990 and then declined from 1990 to 2005. The period of N deposition increase is included in the calculations of impact.

For each ecosystem service, an impact pathway was constructed, deriving response functions based on published literature for each step of the chain. Nitrogen emissions data were from UK emissions inventories (Murrells et al., 2010); deposition data were from emission and interpolation outputs of the FRAME model, spatially calibrated to measured deposition data, summarised in ROTAP (2012). Dose response functions linked the quantity of N deposited to impacts on ecosystem services. Calculations used UK average N deposition data, for forest and non-forest habitats as appropriate, accounting for differences in gaseous N deposition velocities. For each scenario, the difference in pollutant deposition and consequent impact on ecosystem services was calculated for each year, relative to the baseline of no change in deposition. Economic values of ecosystem service provision were calculated using market prices, cost-based approaches and value transfer methods as appropriate. The difference in value between the scenarios for each year was calculated, then Net Present Value using a 3.5% discount rate following UK government guidelines for ecosystem service valuation (HM Treasury, 2003), and Equivalent Annual Value (EAV) using an annuity over the scenario period were calculated. The derivation of response functions for each service and valuation methods are described separately below. Uncertainty analysis for each step of the impact pathway in the valuation was conducted following a Monte Carlo based approach using @RISK (Palisade Corporation, USA). Uncertainties are expressed as lower bound and upper bound 95% Confidence Intervals around a best estimate. The quantification and valuation methods for the six services are summarised below, with more detail provided in the Supplementary Material for some services.

7.2. Impacts of N on timber production

In general in the UK, declines in N deposition lead to reduced tree growth. More detailed description of these assumptions can be found in the Supplementary Material. Ratios of carbon sequestered in tree biomass per unit of deposited N were calculated as 2.3–16 kg C per kg N for decreases in N deposition. Carbon sequestered was converted to dry weight timber assuming 50% C content (Solberg et al., 2009), and then to greenwood volume assuming a representative wood density of 0.45 t dry wood/m^3 for softwood (range 0.33–0.45 t/m^3) and hardwood (0.49–0.53 t/m^3) (Broadmeadow and Matthews, 2003). Wood volume was scaled to the area of forestry under softwood and hardwood. Valuation was based on the standing sales prices for UK forest estate timber, of £120 per hectare for softwood and £30 per hectare for hard woods (at 2010 prices).

7.3. Impacts of N on livestock production (meat and dairy)

Impact of N deposition on livestock production is assessed here via inferred effects of N on the productivity of grassland, and consequently meat (cattle and sheep) and dairy production. Reductions in N deposition are assumed to translate to a reduction in grassland productivity which is compensated by farmers with additional N fertiliser inputs on a 1:1 basis (Smart et al., 2011). The assessment is restricted to improved grassland, although arguably, effects of declining N deposition may be more noticeable in unimproved grassland, valuation of which would require detailed response functions for changes in meat and wool production based on changes in forage quantity and quality. Valuation is based on fertiliser prices of £0.62 per kg N in ammonium nitrate straights (Nix, 2011).

7.4. Impacts of N on carbon sequestration

Quantification of impacts on carbon sequestration was possible for woodland and heathland. There was insufficient information to calculate impacts in grassland. Impacts were based on changes in C sequestration in soils for woodland, heathland, and additionally in above-ground biomass in woodland, following the methodology described in Section 7.2 for N impacts on tree growth. In woodland, the ratio of carbon sequestrated in soils and vegetation per unit of deposited N ranged from 3 to 30 kg C per kg N for decreases in N deposition (De Vries et al., 2009). In heathland, the ratio of carbon sequestrated in soils per unit of deposited N ranged from 3 to 14 kg C per kg N for decreases in N deposition (De Vries et al., 2009). For both habitats, values were adjusted in the context of declining N deposition to take account of lag effects on plant growth due to accumulated N in soils, following Wamelink et al., (2009a,b). Greenhouse gas fluxes were scaled by habitat area to give gross fluxes at the UK scale, and converted to CO_2 equivalents (CO_2e). Valuation was based on the non-traded shadow price of carbon (2010 £51.70/t CO_2e) (DECC, 2010).

7.5. Impacts of N on N_2O emissions

Nitrogen impacts on N_2O emissions were calculated for the following semi-natural habitats: woodland, heathland, grassland, bogs, wetlands, assuming that for each kg of deposited N, 1% is re-emitted into the atmosphere as N_2O–N (IPCC 2006). Greenhouse gas fluxes were scaled by habitat area to give gross fluxes at the UK scale, and converted to CO_2 equivalents (CO_2e). Valuation was based on the non-traded shadow price of carbon (2010 £51.70/t CO_2e) (DECC, 2010).

7.6. Impacts of N on recreational fishing

Impacts of N on recreational fishing occur via changes in nutrient status affecting fish populations and species composition, and indirectly via acidification impacts. This partial valuation estimate is based on changes in eutrophication due to N and subsequent impacts on salmonid fishing in upland rivers. We make the assumption that in these catchments, atmospheric N is the primary source of anthropogenic N, and streamwater nitrate concentrations alter proportionally with changes in deposition. The assumptions behind the contribution of atmospheric deposition to stream nitrates in upland and lowland rivers are provided in more detail in the Supplementary Material.

Since most valuation studies of eutrophication impacts do not separate the contribution of N and P, for this study valuation was based on a travel cost method and a random utility model (Johnstone and Markandya, 2006). Johnstone and Markandya (2006) examined the impact of marginal changes in river water quality for selected rivers in England (a total of 303 stretches of river covering upland, lowland and chalk streams). Their analysis includes the estimation of a trip prediction (participation) model which examines the relationship between river quality attributes and proportional changes in fishing trips, where a 10% increase in nitrate results in a 7% reduction in predicted fishing trips. Johnstone and Markandya (2006) estimate that a 10% increase in nitrate reduces consumer surplus by £1.79 per trip (2001 £). This equates to £2.18 in 2010 prices.

We proxied stream type by fish species with the assumption that all salmonid fishing occurs in upland river catchments. The number of angler days per year for salmon and sea trout angling is 3.8 million, mostly in rivers (Radford et al., 2007). One angler day was equated to one trip (Johnstone and Markandya, 2006). The change in consumer surplus was multiplied by the change in the number of trips calculated to generate a value for the impact of N deposition on recreational fishing in upland rivers. The relationship between the consumer surplus value and the change in trips was assumed to be linear.

7.7. Impacts of N on appreciation of biodiversity

This study defines the ‘appreciation of biodiversity’ in terms of non-use values associated with conserving elements of the natural environment, plant and animal species. The study focuses on biodiversity in terrestrial habitats, using nitrogen critical load exceedance (a damage threshold) as a proxy for habitat damage caused by N deposition (Bobbink et al., 2010). Impacts were calculated for four broad habitat types: woodland, heathland, acid grassland, calcareous grassland and bogs. More detail on the methodology is provided in the Supplementary Material.

Non-use values were based on a choice experiment methodology (Christie et al., 2010; Christie and Rayment, 2012), designed to value changes in the level of provision of seven separate ecosystem service attributes. These ecosystem service attributes included non-charismatic species, defined as follows (Christie et al., 2010): threatened trees, plants, insects and bug species and populations that will be influenced by UK Biodiversity Action Plan implementation, framed in terms of increase or of decrease in the number of species. The WTP value for non-charismatic species was £88 per household.

This WTP value was disaggregated by habitat according to an expert derived matrix, allowing separate attribution of ecosystem service value to the habitats modelled in this study (Table S1, Supplementary Material), comprising eight of the 19 habitats in Christie et al. (2010). The disaggregated WTP values for each habitat were then scaled according to the proportional change in critical load exceedance for that habitat (Figure S2, Supplementary Material). The disaggregated, scaled value was then multiplied by the number of households in the UK, 25 million, to calculate a total value of this service for the UK for each habitat at a given level of historical N deposition.

8. Results

For the six ecosystem services that were assessed, changes in value due to declines in N deposition are shown in Table 1. There was a net decline in average UK N deposition of 2.53 kg N ha^{-1} yr^{-1} from 20.2 kg N ha^{-1} yr^{-1} in 1987 to 17.6 kg N ha^{-1} yr^{-1} in 2005. This resulted in a loss of ecosystem service value for timber production, livestock production, and carbon sequestration, but a gain in ecosystem service value for emissions of the greenhouse gas nitrous oxide, recreational fishing and biodiversity. There was a net benefit for ecosystem services of declining nitrogen deposition: the net equivalent annual value (EAV) of changes in ecosystem services due to changes in pollution deposition was £65.8 m (£5.1 m to £123.2 m, 95% CI) per year. When broken down by ecosystem service category, there was a cost for the two provisioning services of −£6.2 m (−£3.5 m to −£9.2 m, 95% CI) due to a loss of N-stimulated productivity. Regulating services showed a mix of benefit and cost, but amounted to a net loss of −£15.7 m (−£4.5 m to −£30.6 m, 95% CI)). However, benefits to the two cultural services amounted to £87.7 m (£13.1 m to £163.0 m, 95% CI), which outweighed the net costs to provisioning and regulating services.

9. Discussion

In this paper we present an evidence-based summary of nitrogen impacts on ecosystem functions and processes within the context of impacts on ecosystem services. We acknowledge that this is, of necessity, a snapshot of the extensive nitrogen-effects literature spanning many decades. However, it serves to identify where evidence does exist to support assumed impacts on ecosystem services, and where the evidence is lacking, or where there is not yet consensus. Two examples illustrate these points. In both cases, an understanding of the underlying processes helps explain some of the discrepancies. It is usually assumed that N increases productivity of semi-natural grasslands, however productivity of oligotrophic calcareous grasslands and some upland grasslands is often restricted by phosphorus limitation and by climatic factors respectively (e.g. Phoenix et al., 2003). It is also assumed that N increases soil carbon accumulation as a result of stimulating plant productivity. However, accumulation of soil carbon is dependent on the delicate balance between increased carbon inputs to soil from litter and rhizodeposition and increased soil respiration rates. These relationships between soil carbon and nitrogen deposition differ by habitat, and may show contrasting impacts, with N increasing soil C in woodlands but decreasing it in bogs for example (Bragazza et al., 2006; De Vries et al., 2009).

With the evidence so far, it is possible to quantify impact pathways for N impacts on selected ecosystem services, but it should be noted that they entail a range of assumptions. These include the issue of time lags and the reversibility of N impacts following reductions in deposition. Because N alters rates of

Table 1
Summary of costs (negative) and benefits (positive) to ecosystem services as a result of declines in nitrogen emissions (£million Equivalent Annual Value – EAV, 95% CI in brackets), by ecosystem service.

Provisioning services		***Regulating services***		***Cultural services***		**Net EAV**
Timber production	**Livestock**	**Net GHG emissions**		**Recreational fishing**	**Appreciation of biodiversity**	
		CO_2	N_2O			
Loss −£1.80 (−£0.7– −£3.5)	Loss −£4.40 (−£2.8– −£5.7)	Loss −£21.00 (−£7.2– −£39)	Gain £5.30 (£2.7–£8.4)	Gain £0.03 (No uncertainty estimate performed)	Gain £87.70 (£13.1–£163)	Gain £65.80 (£5.1–£123.2)

environmental processes, and accumulates in ecosystems, its effects are often long-lived. A reduction in deposition does not remove N from the soil–plant system. The continued high rates of tree growth (Wamelink et al., 2009a) reflect the ongoing impacts of N stored within soils. Active intervention is required to remove N from soils, and even then hysteresis effects e.g. due to associated acidification of soils or waters mean that recovery is slow, may not take the same trajectory, or indeed may be irreversible. This concept of irreversible change is intimately linked with the idea that an ecological threshold should not be exceeded, i.e. with the idea of a critical load (Smart et al., 2011).

Additional assumptions are applied in the quantification of the full impact pathway including the valuation steps. For example, it is assumed that farmers notice changes in grassland productivity associated with declining N deposition, and adjust their management accordingly. This is a strong assumption, and farmer decisions are not governed solely by productivity. In intensively managed systems where fertiliser application rates are high relative to deposition this may not be the case, yet when changes in deposition are large, as for sulphur, effects on yield are noticeable (Zhao et al., 2003) and farmers change their management accordingly. In this case, it is likely that the assumed response of farmers to changes in N deposition over-estimates the financial impact on livestock production. The validity of valuing biodiversity is much debated, and methods are still in their infancy. Here we use value transfer from a WTP study on biodiversity valuation (Christie et al., 2010; Christie and Rayment, 2012) to a different valuation context, but where the outcome that respondents were asked to value is comparable (a decline in species populations of non-charismatic priority species). Non-charismatic species were chosen rather than the higher valuations attributed to charismatic species in Christie et al. (2010) since the non-charismatic species closer represent the wide range of organisms known to be negatively affected by N deposition. The WTP estimates in Christie et al. (2010) are within the range of other studies valuing biodiversity of multiple species (Nunes and van den Bergh, 2001). Other studies have applied a restoration cost to value nitrogen impacts on biodiversity (Ott et al., 2006) which include a different set of assumptions: that it is possible to restore all impacted areas, that the methods and costs are known, and that all impacted areas are restored. Value transfer remains a reasonable option until studies are specifically designed to value atmospheric N deposition impacts on biodiversity. Although frequently criticised, stated preference techniques remain the only method for capturing non-use values, and many of the criticisms do not apply in well-designed studies (Carson et al., 2001). Ignoring non-use values risks only partially capturing the impact of improvements in air pollution in an ecosystem services framework, which aims to provide a more holistic approach to evaluating policy impacts.

Here, we show through selected examples that it is possible to quantify impact pathways for N deposition using an improved understanding of bio-physical relationships and the evidence-based links to ecosystem service provision. We also illustrate their potential application in a policy context, which suggests considerable net benefit has been obtained as a result of policy measures to reduce N deposition, primarily of oxidised N from vehicle exhausts and large combustion sources, despite costs associated with declines in N-stimulated production in provisioning services. While the approaches here differ from those used in the European Nitrogen Assessment (Sutton et al., 2011b), the relative magnitude of impacts on ecosystems and on climate are similar, with much greater value associated with impacts on ecosystems. The net benefit in this study hinges on capturing the non-use values associated with appreciation of biodiversity in terrestrial habitats. In general, provisioning services which produce saleable commodities are relatively easy to value through market prices. By contrast, cultural services are much more difficult to value, and impacts on aquatic biodiversity, water quality and other recreation-related cultural services, which are all likely to see an increase in the level of service provision as a result of declines in N deposition, are not currently valued. This means that, until the methodologies for valuing cultural services and the valuation evidence base improve, assessments will be weighted in favour of provisioning services which, as shown in this study, may lead to underestimates of the benefits of reducing atmospheric N deposition.

Further development of this work should consider a number of issues. Constructing the impact pathways for ecosystem services requires multi-disciplinary teams involving environmental and social scientists, and environmental economists. In this study we present broad-brush assessments at national scale in order to show the broad applicability of the approach, but they can be further improved by more detailed spatial quantification of environmental impact and of valuation, as has been shown for ozone impacts on selected provisioning services (Mills et al., 2011; Karlsson et al., 2005). Nevertheless, there remain knowledge gaps in the underpinning environmental science, knowledge gaps in valuation, and in the impact pathways linking them, particularly for cultural services. These knowledge gaps need to be addressed before many of the impacts of N on other ecosystem services can be valued.

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Appendix A. Supplementary materials

Supplementary data associated with this article can be found in the online version at http://dx.doi.org/10.1016/j.ecoser.2013.09.001.

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