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Applying the ecosystem service concept to air quality management in the UK: a case study for ammonia[†]

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To date evaluation of the benefits of policies to control emissions of air pollutants in the UK has focused on human health effects, which are quantified economically, whereas ecosystem protection has been assessed using critical levels and critical loads. This paper considers the current feasibility of using an ecosystem services approach to appraise the benefits of alternative scenarios for controlling agricultural ammonia emissions in the UK. The effect of ammonia emission reductions on ecosystem service delivery was assessed using an impact pathway approach. A 'weakest link' analysis identified that economic valuation of impacts on many key ecosystem services was constrained by inadequate dose–response relationships to predict physical changes in service flows and/or by an inability to produce economic valuations of the predicted physical changes. For effects on biodiversity, both the timescale of response and poorly defined relationships between changes in species composition and ecosystem service delivery are significant barriers. However, it was possible to produce indicative values for the marginal impact of ammonia abatement measures on climate regulation; the values obtained were comparable in magnitude to those for human health impacts. The ecosystem service approach thus offers the potential to provide a holistic appraisal of the effects of emission reductions, and could therefore make a valuable contribution to future air quality management. However, improvements in data collection and quantification methods are needed before a full ecosystem services-based evaluation of costs and benefits becomes possible for ammonia and for other major air pollutants.

Keywords: nitrogen; ammonia; economic valuation; ecosystem service; air pollution; biodiversity; climate regulation

1. INTRODUCTION

Emissions of ammonia (NH_3) to the atmosphere can have significant effects on a range of sensitive terrestrial and aquatic ecosystems, through increased deposition of nitrogen causing eutrophication and acidification, and on human health through the formation of secondary inorganic aerosols. Ammonia management policies in the UK have been driven primarily by the requirement to comply with the Gothenburg Protocol of the United Nations Economic Commission for Europe (UNECE) Convention on Long-range Transboundary Air Pollution (CLRTAP) and the EU National Emissions Ceilings Directive (NECD), which committed the UK to a national emissions target for 2010 of 297 kt NH_3 per year. The CLRTAP Gothenburg Protocol targets are currently being re-negotiated and are likely to require lower emissions ceilings and mandatory measures for NH_3, to be met by 2020 (Amann *et al*., 2010; UNECE, 2010). This has led the UK government to consider various policy options to further reduce NH_3 emissions. Potential emission control scenarios have been investigated and indicative estimates of the costs which such scenarios would incur have been produced (Misselbrook, 2007).

It is important for policy evaluation to assess the benefits, as well as the costs, of investments in different options for reducing NH_3 emissions. To date, these benefits for human health and for the natural environment have been assessed by quite different approaches. The

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impacts of NH_3 emissions on health arise primarily through aerosol effects on respiratory and cardiac disease. Economic assessment of the health benefits of a given reduction in emissions follows standard procedures developed for air pollution by the UK government (Defra, 2006), which predict the reduction in mortality and morbidity nationally, and then apply standard valuations of reductions in years of life lost and improvements in health quality to produce an economic valuation of those mortality and morbidity reductions.

A different approach has been used historically to assess the ecological benefits of reductions in NH_3 emissions. Critical thresholds have been defined, above which adverse effects can occur: either critical levels for gaseous NH_3 (Sutton *et al*., 2009) or critical loads of total nitrogen deposition (N_{dep}), to which NH_3 emissions along with those of nitrogen oxides (NO_x) contribute (Bobbink *et al*., 2003, 2010). National evaluation has primarily been based on critical loads for total N_{dep}. It is projected that currently 60% of the total UK area of sensitive habitats receive N_{dep} above their critical loads, and that this is expected to decrease to 49% by 2020 (RoTAP, 2011) due to reduced emissions of NH_3 and NOx, thus providing a yardstick against which the impact of further NH_3 emissions reductions on ecosystems can be assessed. However, this method only partly evaluates the implications for the natural environment of changes in agricultural policies and practices to reduce NH_3 emissions, because these may have secondary effects on water quality, greenhouse gas budgets and primary production which are currently not considered when setting critical loads. Furthermore, the degree of critical load exceedance is not directly linked to effects that can readily be quantified and valued.

Hence, a new and more holistic approach is needed if benefits (and dis-benefits) of NH_3 emissions reductions are to be evaluated for both the natural environment and human health. The concept of ecosystem services provides such a holistic framework for informing management of ecological-social systems allowing, at least in principle, the evaluation of the full costs and benefits of potential policy options (MEA, 2005; Defra, 2007a). Central to this approach is the concept that natural ecosystems provide a wide range of services to society (e.g. flood control, crop pollination, water purification, recreation and crop provisioning) and that economic valuations of the relative value of policy-induced changes in the delivery of these different services should be taken into account when environmental policy is being designed.

In addition to, and typically quite distinct from, these ecosystem service dimensions are the direct effects which air pollutants can have on human health and on physical structures such as buildings. An ecosystem services-based valuation could, in principle, provide a mechanism for quantifying the economic consequences of changes in ecosystem service delivery resulting from further reductions in NH_3 emissions, which could then be incorporated alongside valuation of the beneficial effects on human health within an overarching cost-benefit analysis.

However, the practical difficulties of applying an ecosystem services approach for controlling NH_3 emissions, and to air quality management in general, have to date not been systematically evaluated. This paper seeks to assess the feasibility of applying an ecosystem services-based assessment to air quality management; in particular, we assess the methodological obstacles encountered when applying this approach within an agricultural NH_3 emissions context in the UK. The potential advantages of an ecosystem services-based assessment for agricultural NH_3 management include: (a) explicit recognition of the links between NH_3 emissions, environmental quality and the full suite of economic values which ecosystem services deliver, and (b) identification of synergies, antagonistic effects and tradeoffs among the outcomes produced by any proposed policy change. This paper, therefore, considers the current practicality of an ecosystem services-based assessment which aims to deliver these potential advantages. We draw throughout on a detailed description of our ecosystem services-based assessment of agricultural NH_3 management (Hicks *et al*., 2008), to which the reader is referred for additional detail and explanation. A comprehensive evaluation for all policy options, habitats and ecosystem services is outside the scope of this paper, and we therefore focus in detail only on attempts to apply an ecosystem services-based valuation to quantify a limited number of the potential benefits of reduced NH_3 emissions.

We begin by providing a qualitative overview of NH_3 emission effects on ecosystem services in the UK, and then outline the steps needed to quantify the effects of emission reductions on delivery of these services. A ‘weakest link’ analysis is used to assess whether the relevant impact pathways (see Defra, 2007a) can currently produce quantified predictions of changes in the value of the services provided by terrestrial ecosystems following reductions in NH_3 emissions. Impact pathway-derived valuations for changes in the delivery of two ecosystem services, biodiversity provision and climate regulation, are then considered in more detail and compared to an indicative valuation of the benefits of NH_3 emissions reduction for human health. We then consider the potential which the ecosystem services framework offers for holistic assessment of the environmental impacts of proposed policy changes and discuss the challenges currently posed by full implementation of such a framework. Finally, we consider whether the feasibility and reliability of ecosystem services-based assessment could be improved by modifying data collection techniques, dose–response modelling, environmental valuation methodologies and/or by improving connectivity along the impact pathways through which changes in ecosystem service delivery are quantified and valued.

2. OVERVIEW OF CHANGES IN ECOSYSTEM SERVICE DELIVERY

2.1. Initial qualitative assessment

To provide a starting point for analysis, major potential effects of N_{dep} were listed by the ecosystem service that is affected (Table 1). Supporting services are not considered in this analysis to avoid the risk of double counting. The effects listed in Table 1 are not associated with any particular level of NH_3 emissions, but merely identify the types of linkage that exist to ecosystem service delivery. This assessment was based on a recent expert review of the impacts of air pollutants in the UK, which included an overview of effects on ecosystem services (RoTAP, 2011). It is primarily based on evidence of total N_{dep} effects, rather than the specific effects of NH_3. RoTAP (2011) concluded that reduced nitrogen deposition, to which NH_3 emissions contribute, is likely to have even more severe effects on ecosystems than oxidised nitrogen deposition, but that such differential effects are expected to vary according to the ecosystem services being considered. Therefore, for the purpose of this paper with a UK focus, we assumed that the effects of oxidised and reduced N_{dep} on ecosystem services are equivalent, although this may provide a conservative indication of NH_3 emission effects.

Table 1. Effects of nitrogen deposition (N_{dep}) on ecosystem services in the UK (based on a qualitative summary in RoTAP (2011)). Broad effects are indicated in normal font with specific examples of differences in response of different processes or habitats indicated in italics

Ecosystem service category	Effect of NH_3 emissions
1. Provisioning Services	
Ecosystem goods	Production of goods (e.g. food, fuel, fibre) can be increased and decreased *Salmonid fish populations can be reduced in acidified freshwaters* *Moderate increases in N_{dep} can increase forest growth rates*
Water quality	Acidification and eutrophication of surface waters can be caused by direct deposition or by leaching from terrestrial ecosystems
Biochemical/genetics	Number and diversity of species can be reduced in both terrestrial and aquatic ecosystems
2. Regulating services	
Air-quality regulation	Air quality can both increase and decrease *N_{dep} increases tree growth, improving removal of air pollutants* *NH_3 emissions contribute to formation of secondary particulates which affect health*
Climate regulation	Greenhouse gas fluxes can both increase and decrease *Carbon sequestration in soils and vegetation can increase in N limited sites or decrease if critical loads are exceeded* *Nitrous oxide production can increase with increasing N_{dep}*
Water regulation	Increased rates of peat creation and forest growth can increase water storage and interception
Natural hazard regulation	Inadequate evidence of effects overall, but increases in vegetation cover can increase erosion regulation *Erosion effect especially important in coastal dune habitats*
Pest and disease regulation	Inadequate evidence of effects
Pollination	Pollination can be both increased and decreased *N_{dep} increases flowering in heathlands* *N_{dep} decreases flowering in grasslands*
3. Cultural services	
Recreation and tourism	Changes in terrestrial and aquatic species composition may affect field sports and ecotourism
Aesthetic	Changes from the status quo (e.g. changes in species composition) may be judged to be significant
Educational	Reduction in species rich habitats as sites for study
Cultural heritage	Loss of iconic species

For some of the ecosystem services listed in Table 1 (e.g. primary production) there is a strong evidence base for the effects of NH_3 emissions, while for others (e.g. pest and disease regulation) the evidence is not adequate to identify a consistent effect of NH_3 emissions. Some of the effects on service delivery arise from general responses across several habitats. For example, fertilisation effects from additional N input will increase harvestable material (e.g. timber (woodlands), hay (semi-natural grasslands)) while nitrate leaching in runoff entering water bodies from different terrestrial habitats will reduce water quality and can accelerate eutrophication and affect biodiversity. The size of these effects will vary significantly between habitats, depending on the degree to which production is limited by nitrogen. In the case of erosion regulation, for example, there are specific effects in coastal dune habitats, because additional N_{dep} increases the growth of 'sand holding' grass and sedge species. For several services, contrasting effects were identified, primarily because increased N_{dep} has different effects on different processes (e.g. for CO_2 and N_2O) fluxes and hence climate regulation) and habitats (e.g. different responses of flowering and hence pollination in heathlands and grasslands). This complexity could be accommodated by extending Table 1 to include a habitat-specific analysis. Such an extension is outside the scope of this paper, but is discussed by Hicks *et al*. (2008).

A further important factor in evaluating both the baseline for service delivery before further policy intervention, and the effects of different policy options, is that NH_3 emissions from different types of sources in UK agriculture may have contrasting impacts. For example, a more detailed analysis would consider the differential effects of point sources (e.g. intensive pig and poultry units) versus diffuse sources (e.g. fertiliser and manure applications), and of constant versus intermittent sources. Such a detailed analysis is also outside the scope of this paper, but is discussed further by Hicks *et al*. (2008).

Based on the ecosystem service effects categorised in Table 1, further NH_3 abatement measures would be likely to produce improvements in the delivery of ecosystem services reliant upon air and water quality regulation, biochemical/genetic services and climate regulation, the latter as a consequence of decreased N_2O emissions. However, reductions in ecosystem service delivery could result for harvested goods such as timber, and for climate regulation as a consequence of reduced carbon sequestration.

2.2. 'Weakest link' analysis to determine which ecosystem service changes can be valued quantitatively

The next step in an ecosystem service-based assessment is to construct an impact pathway to predict the quantified physical changes in the delivery of particular ecosystem services under the various abatement scenarios, and then to estimate the economic value of those changes. This quantification and valuation would, in theory, proceed by:

1. applying the proposed emissions reductions to spatially located NH_3 sources;
2. using appropriate models of air chemistry and air transport to predict consequent changes in the distribution of NH_3 concentrations and NH_x deposition;
3. quantifying spatial deposition changes for different key habitats;
4. applying relevant dose–response relationships to predict changes in ecosystem function and subsequent changes in ecosystem service delivery within particular habitats, and
5. applying relevant valuation functions, site-specific where necessary, to estimate the changes in social value which arise from changes in ecosystem service delivery.

The first three of these stages are standard methods which would be applied in any UK-wide evaluation for air quality management, including those using the critical load approach (RoTAP, 2011). However, the fourth and fifth stages of the impact pathway raise the following key questions regarding the feasibility of ecosystem services-based assessment: is there sufficient capability under current knowledge to make robust and reliable quantified predictions of changes in (i) ecosystem condition and function by habitat type, (ii) ecosystem service delivery and (iii) social value under the different abatement scenarios? We use a 'weakest link' approach to address this issue.

Knowledge and data gaps inevitably appear in many of the impact pathways linking changes in NH_3 emissions to changes in the social value of ecosystem service delivery. It is, therefore, essential to determine, as objectively as possible, whether each of the impact pathways which link the various abatement measures through their environmental consequences to an economic valuation of a change in the delivery of a particular ecosystem service, is amenable to quantitative valuation. The validity of the decisions made at this stage dictates the scope, reliability and completeness of the final ecosystem service-based assessment. For policy application, detailed stakeholder consultation might be expected to prioritise the effects in Table 1 and assess whether impact pathways can be constructed for each of these.

A full evaluation of all the effects of reductions of NH_3 emissions on ecosystem services is outside the scope of this paper. In order to illustrate the differing nature of the challenges involved, we focus on two effects which were identified as being of high policy significance by RoTAP (2011). Figure 1 shows schematic impact pathways for policy-induced changes in the provision of generic ecosystem services and greenhouse gas emissions, and compares these with the impact pathway by which human health outcomes are valued. The initial stage in each of these impact pathways requires spatially specific dose–response relationships to link proposed abatement measures to physical changes in ecosystem service delivery, human health outcomes or greenhouse gas emissions. The pathways then differ, however, in their requirement for spatially specific valuation of these predicted physical changes. The economic consequences of physical changes in human health and

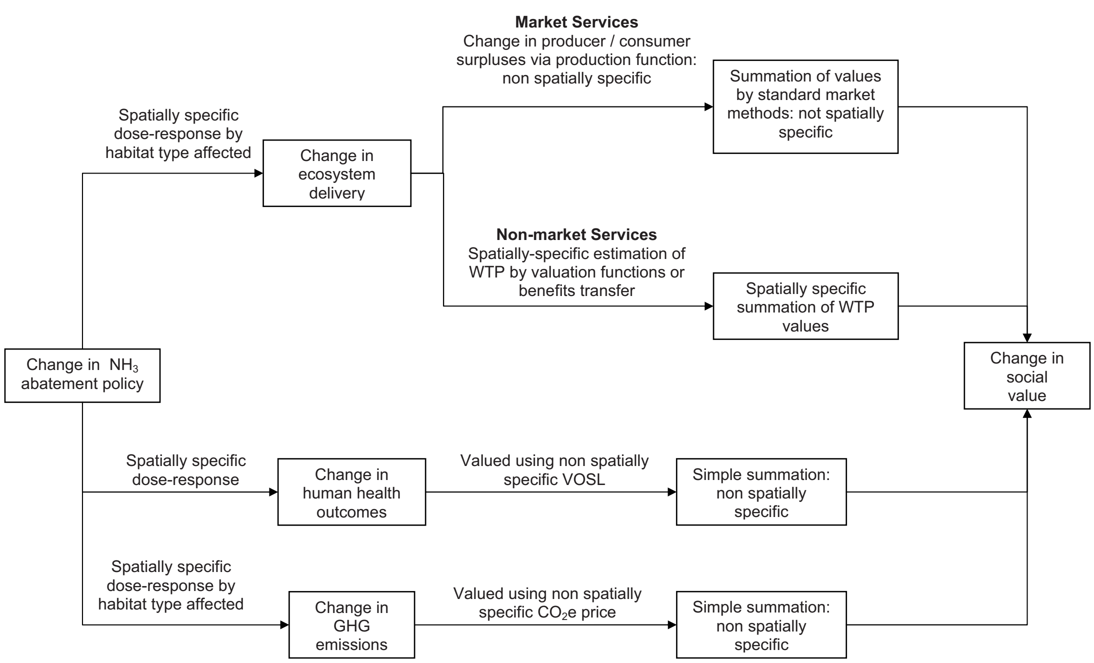

Figure 1. Contrasting requirements for spatially specific dose–response relationships, valuation methods and value summation for changes in ecosystem service delivery, human health outcomes and greenhouse gas emissions following a change in NH_3 abatement policy. GHG, greenhouse gas; WTP, willingness to pay; CO_2e, carbon dioxide equivalent; VOSL, value of a statistical life

greenhouse gas outcomes can be valued by established, non-spatially specific methods (see Section 3), whereas the economic value of changes in several other ecosystem service flows—notably non-market cultural, recreational and heritage services—will typically have to be spatially specific to produce meaningful results. Quantification of the economic value of changes in these service flows is therefore more challenging because benefits transfer will usually be required to derive the necessary site-specific valuations within the limited budgets which are typically allocated for policy appraisal. Benefits transfer is the process by which an estimate of the social value of a marginal change in ecosystem service delivery at one site is produced from prior estimates of the social value of marginal changes in delivery of the same service at another site or sites. Defra (2007a) provides a fuller description of the categories of total economic value (use value, non-use value etc.) delivered by different types of ecosystem service, together with a brief description of the various valuation methodologies used to estimate changes in those different categories of value (revealed vs. stated preference methods).

Table 2 summarises our weakest link analysis. The reliability assessments in Table 2 for individual links in the pathways are based on a literature evaluation by the authors, with their combined expertise in relevant scientific and economic research. More extensive consultations or assessment would help to further refine our evaluation. In our 'weakest link' analysis, quantified valuation of a change in ecosystem service delivery was considered to be fully reliable, and was thus awarded a *** valuation rating, if it could produce a valuation result for which:

(a) dose–response functions could specify the actual changes in ecosystem function and condition, and the consequent physical changes in service flow outputs, with adequate accuracy, precision and spatial specificity;
(b) a well-established valuation methodology was available to quantify the economic value of the physical changes in service flow output; and
(c) the necessary valuation functions and results were available to implement benefits transfer for site-specific economic valuation of physical changes in service flow output, where relevant.

On this basis, none of the ecosystem service impact pathways can be placed in the highest overall reliability class. The relative strictness of the scoring system in our 'weakest link' assessment can be judged against the ** overall reliability rating assigned to the impact pathway currently used by the UK National Air Quality Strategy (NAQS) for reductions in human morbidity and mortality. The 'weakest link' in this pathway was judged to be the dose–response relationship between NH_3 emission control measures, $PM_{2.5}$ aerosol concentrations and human health outcomes (see Defra, 2007b).

In the following sections we focus on a more detailed discussion of the challenges inherent in producing a quantified valuation of changes in biodiversity provision and climate regulation (assessed at 0 and * reliability, respectively, in Table 2). This illustrates the type of issues that currently impede full quantification by ecosystem services-based assessment. These issues are then compared with those inherent in valuing changes in human health outcomes, for which the challenges to reliability are deemed acceptable for use in policy evaluation by the UK government.

3. ECOSYSTEM SERVICE VALUATION FOR BIODIVERSITY AND CLIMATE REGULATION

3.1. Valuing service flow changes driven by terrestrial biodiversity

Biodiversity is not an ecosystem service in itself, but aspects of terrestrial biodiversity such as species diversity/species richness or species composition affect delivery of a number of different ecosystem services. The role of biodiversity in supporting, or directly delivering, these various ecosystem services could, in principle, be valued using relevant impact pathways of the form illustrated in Figure 1. The first step in all of these pathways is the exposure–response relationship between N_{dep} and species diversity or species richness.

Dose–response relationships between N_{dep} and plant species richness (number of species per unit area) have been defined for sensitive grassland and other ecosystems based on results from experimental manipulations (e.g. Bobbink *et al*., 2010) and from field surveys (e.g. Stevens *et al*., 2004; Maskell *et al*., 2010). In principle, such relationships could be used to estimate the number of species that would be restored following a given reduction in NH_3 emissions. However, a number of difficulties arise even in the application of these apparently simple relationships.

Firstly, the slopes of the relationships are quite different for the experimental and field survey data. The relatively low slope for the experimental studies may partly reflect the relatively short duration of many of these studies compared to the decades of elevated N_{dep} which have been received in most areas of the UK—hence, they do not capture the full long-term impacts of any change in NH_3 emissions. In contrast, the field surveys provide a relationship between species richness and current N_{dep}, but it is likely that variations in current estimates of N_{dep} are surrogates for the cumulative N_{dep} over a number of decades. In this case, it is uncertain how to quantify the impacts of changes in NH_3 emissions, since the observed effects are the result of cumulative deposition over a long, but unknown, period of time. Even if the timescale was known, it is far from certain that the timescale of recovery (as NH_3 emissions decrease) will be the same, especially if species have been completely lost from an area.

Secondly, interpretation of how dose–response relationships for species richness affect ecosystem services depends on whether species are of value because of their rarity, their cultural value, or their role in providing or supporting particular ecosystem services. In the case of Stevens *et al*. (2004), for example, six species were identified as being consistently lost at sites with high rates of N_{dep}. These included three forbs *Plantago lanceolata* (ribwort plantain), *Campanula rotundifolia* (harebell) and *Euphrasia officinalis* (eyebright); the grass *Molinia caerulea* (purple moor grass); the shrub *Calluna vulgaris* (heather) and the moss *Hylocomium splendens* (mountain fern moss). Each of these might be associated with particular ecosystem services and/or contribute value directly in their own right. In some habitats, species lost as N_{dep} increases are replaced by other species so that overall species richness does not change (Bobbink *et al*., 2010); here the contribution to ecosystem services of both the lost species and their replacements would need to be considered.

Table 2. Reliability and acceptability of different stages in the impact pathways used to value changes in ecosystem service delivery from terrestrial habitats following changes in NH_3 abatement policies

Benefit/disbenefit via particular ecosystem service(s)	Valuation methodology	Reliability of dose/response function(s)	Acceptance of valuation methods	Reliability of local valuations driven by 'response' from science stages	Reliability of spatial summation of local valuations	Overall Reliability (following 'weakest link' approach)
Land-based recreation and tourism	Site-specific dose/response followed by site-specific valuation	Link to species richness and composition only*	Travel cost, contingent valuation, choice experiments***	Valuations of WTP for changes in recreational experience do not respond to species diversity*	Benefit transfer required for site-specific valuations: unlikely to be able to control valuations for spatial variation in rarity*	*
Changes in availability of biochemical/genetic resources from terrestrial species	Site-specific dose/response followed by estimation of option value & quasi-option value	Link to species richness and composition only*	Contingent valuation, choice experiments**	unknown	None available 0	0
Changes in climate regulation (carbon sequestration, GHG emissions, pollution swapping)	Site-specific dose/response relationships followed by valuation of changes in GHG emissions using DECC-specified methodology	Controversy surrounds link to carbon sequestration*	Speculation regarding future MAC for carbon**	DECC prescribed methodology using change in GHG emissions; no need for spatially-specific adjustment***	Site-specific dose response followed by simple summation; no need for benefits transfer***	*
Reduction in human morbidity and mortality	Site-specific dose/response relationships between $PM_{2.5}$ aerosol and human health. Changes in health outcomes valued via VOLY methods	Reliable assuming change in $PM_{2.5}$ can be modelled**	Well established methodology & VOLY value***	VOLY approach follows prescribed methodology; no need for spatially-specific adjustment***	Site-specific dose response followed by summation to estimate effects on whole population***	**

GHG is greenhouse gas; WTP is willingness to pay; MAC is marginal abatement cost; VOLY is value of a life year lost, DECC is Department for Environment and Climate Change (UK Government).
Key: 0 denotes missing or not available, through *, ** to ***, where *** denotes a method which is well accepted or believed to be reliable.
A weakest link approach is used to determine reliability of the impact pathway as a whole.

Dose–response relationships are available for some individual species; *Drosera rotundifolia* (sundew) is one such rare species of low nutrient habitats that is very sensitive to increased N_{dep}. This species, through its carnivorous growth habit and long history of use in herbal medicine, receives a high public resonance; it was, for example, voted as the official 'county flower' for Shropshire (Plantlife, 2010), an area with high rates of N_{dep}. In fact, published data from an experimental study on this species (Redbo-Torstensson, 1994) show a clear dose–response relationship with a threshold for declining abundance above an N_{dep} rate which is exceeded over many parts of the UK. Similarly, van den Berg *et al*. (2011) identified a reduction in the number of scarce and rare species by a factor of two in a field survey of calcareous grassland sites when N_{dep} exceeded the critical load range; this loss of rare or scarce species is also likely to link directly to delivery of a cultural heritage service.

Thirdly, the potential for recovery as N_{dep} decreases is uncertain, and some changes may be irreversible. In this case, any level of reduction in NH_3 emissions will not automatically restore the original species complement without active restoration measures, and hence simple relationships between N_{dep} and species richness cannot be used in impact pathway evaluation. 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.

Where estimates of the welfare value associated with the loss, or restoration, of a particular species at a particular location are not available, either directly or derived via reliable application of benefits transfer, then restoration or replacement costs can potentially be used as rough proxies for the social values involved. There is, however, no guarantee that society as a whole would actually have chosen to incur these restoration expenditures, had they been consulted beforehand. The valuations produced by these replacement cost methods are therefore very unlikely to correspond to the true social value of species loss or replacement (Champ *et al*., 2003).

As an example, extensive restoration programmes have been underway in the Netherlands for the last decade to restore valued habitat such as wet and dry heathland and species-rich grassland which has been lost primarily due to elevated NH_3 emissions. Annual investment in restoration programmes from government and EU grants is estimated to be approximately €8M (about £6.4M) (van den Berg, Personal Communication). The costs of restoration programmes of this type targeted at reversing the adverse ecosystem effects of NH_x deposition, could provide very rough proxies for the social value of ecosystem service restoration where relevant species- and/or site-specific valuations are absent and there is insufficient detail in the literature to attempt benefits transfer by meta-analysis.

An additional complication arises because the marginal social value of restoring endemic species at sites adversely affected by NH_3 would be expected to decline rapidly as restoration and recovery proceeds (Defra, 2007a). The social value contributed by re-establishing an endemic at any particular site would therefore decline as endemics re-established themselves at more and more sites across the landscape. Valuation by benefits transfer is difficult in these circumstances because those marginal valuations for species restoration which exist in the literature will have been produced against a given background level of 'rarity' for the species and sites concerned. It is unlikely to be appropriate to transfer a marginal valuation for restoration of particular species at particular sites directly to another context with a different background level of 'rarity'.

Given all of these complications, producing an appropriate valuation for the changes in ecosystem service flows which arise from changes in species composition and abundance following implementation of NH_3 control measures appears likely to present a considerable challenge for some time to come; the 'broken links' here appear in both the 'scientific' and 'valuation' stages of the impact pathways.

3.2. Impact pathway for valuing changes in climate regulation

Quantifying how climate regulation will respond to changes in NH_3 emissions is problematic because a number of separate greenhouse fluxes are affected and the relationships involved are often nonlinear and depend on a wide range of location-specific factors. Decreases in NH_3 emissions could benefit the provision of ecosystem goods from forestry, and increase carbon sequestration, in areas where deposition has been high enough to disrupt nutrient balances and cause forest damage. However, in many areas, tree growth is limited by nitrogen availability, and if NH_3 emissions decrease it is therefore likely that provision of ecosystem goods and carbon sequestration could also decrease. Furthermore, reductions in emissions of N_2O, a potent greenhouse gas, could result from reduced application of animal manures and artificial N fertilisers to agricultural land and reduced N deposition to other habitats.

These conflicting changes are quantifiable in principle, but the complexity and spatially specific nature of the processes involved makes actual quantification of these pollution-swapping tradeoffs challenging under current knowledge. Furthermore, other effects of N_{dep}, for instance, on soil carbon sequestration and methane fluxes, are not well quantified but could be significant.

In contrast to the problems of valuing the net benefits from changes in biodiversity, a straightforward mechanism is available to value changes in climate regulation using the 'per unit' valuation for marginal changes in carbon dioxide equivalent (CO_2e) emissions specified in the UK government's 'target consistent' approach (DECC, 2009). The UK Department of Energy and Climate Change (DECC) recommend that until the envisaged convergence of global carbon pricing in 2030 (DECC, 2009), different 'per unit' marginal valuations should be applied to changes in carbon emissions sourced (i) from major combustion plant covered by the EU emissions trading scheme, and (ii) from sectors of the economy which are currently outside the emissions trading mechanism. Consequently, DECC recommend that changes in carbon emissions from agriculture and changes in carbon sequestration from forests and woodlands in the developed world should be valued using their estimate of the non-traded marginal abatement cost per tonne CO_2e for the year in which those emission changes take effect (DECC, 2009). The valuation which DECC currently recommend should be applied to changes in greenhouse gas emissions from agriculture and forestry which will be enacted in 2020 (the year in which the Misselbrook (2007) abatement scenarios would have taken full effect) is £60 per tonne CO_2e (central estimate in 2009 prices, with a $\pm$50% sensitivity range from £30 to £90 per tonne CO_2e). Indicative valuations of changes in CO_2e emissions (through reduced carbon sequestration or increased N_2O emissions) are produced here on this basis.

Table 3. Illustrative comparison of the *total* cost of suggested NH_3 abatement scenarios and the cost or benefit arising from a *one-off annual* change in the marginal impacts of NH_3 emissions on health and climate regulation outcomes

	Health effect of reduced $PM_{2.5}$[†] (£ Million)[§]	Reduced carbon sequestration[†] (£ Million)[§]	Reduced N_2O emissions[†] (£ Million)[§]	Implementation of abatement scenario[‡] (£ Million)[§]
Scenario 1 (~47 kt reduction)*	+86	−52	+44	−165
Scenario 2 (~67 kt reduction)*	+124	−74	+63	−224
Scenario 3 (~87 kt reduction)*	+160	−96	+81	−362

+ denotes benefit to society, − denotes cost to society.
*Reduction relative to 2010 NEC target of 297 kt emissions total.
†Estimated costs and benefits from the specified *one-off annual* reduction in emissions.
‡Total implementation cost estimated by Misselbrook (2007).
§Values quoted in 2010 present value using 2009 £.

3.3. Indicative calculation of the benefits to climate regulation

Given the increasing importance of climate regulation in UK environmental policy, we have made a first estimate of the value which policies to reduce NH_3 emissions will deliver for climate regulation. Three policy scenarios for sequential emissions reductions were identified, relative to an emissions baseline of the existing NECD 2010 target of 297 kt of NH_3 emissions. Misselbrook (2007) identified the following packages of measures as the most cost-effective ways of achieving different national NH_3 emission ceilings for 2020. Scenario 1 aimed to achieve a 250 kt emissions ceiling (47 kt reduction in the 2010 emissions level) through policy measures targeting fertilisers, land spreading of animal manures, modifications to pig and poultry housing and slurry storage. Scenario 2 aimed to reduce emissions to a 230 kt target (67 kt reduction on 2010) through more stringent measures on slurry storage. Scenario 3 aimed at a 210 kt (87 kt reduction in 2010) target by further tightening regulations relating to pig and poultry housing (Table 3).

To facilitate comparison with the costs of implementing Scenario 1, 2 and 3 abatement measures, the net benefits of emissions reductions which arise in 2020 have been discounted back to their equivalent present values in the 2010 timeframe: the timeframe in which it is assumed that the abatement costs will have been incurred. HM Treasury's standard 3.5% annual discount rate was used (HM Treasury, 2003).

Although there is considerable uncertainty, the best estimates of the increase in above-ground carbon sequestration in European forests and woodlands due to the increased N_{dep} over recent decades vary between 30 and 70 kg C per kg deposited N (Sutton *et al*., 2008). As was the case for biodiversity, we cannot assume that a particular decrease in N_{dep} will cause the same decrease in carbon sequestration, and the timescale involved is also uncertain because tree growth responds to soil N availability, not N_{dep}. However, assuming that the same relationship would apply to reduced N_{dep} in 2020, taking a central annual estimate of 50 kg ha^{-1} of carbon sequestered per kg ha^{-1} of N deposited to woodland annually, and knowing that one tonne of carbon sequestered is equivalent to 3.67 t of CO_2e, this is equivalent to a reduction of 183.5 t of CO_2e sequestered per hectare of woodland annually for each tonne annual reduction in deposited N. Applying DECC's recommended £60 per tonne CO_2e valuation to this reduction in carbon sequestration enacted in the year 2020 produces a social cost estimate of approximately £11 000 for a one tonne reduction in N_{dep} which takes effect in 2020, with an equivalent (2010) present value of £7 800 per tonne N_{dep} reduction after discounting at HM Treasury's recommended 3.5% per annum.

A modelled relationship between spatially explicit changes in NH_3 emissions and N deposition to UK woodlands, taking account of the spatial variation in forest growth rates, is required to expand this valuation to a national scale. However, the likely magnitude of the value of the reduction in carbon sequestration in forests and woodlands nationally can be estimated, for example for Scenario 1, by assuming (i) that all 47 kt of Scenario 1 NH_3 abatement are converted to reductions in N_{dep} within the UK (this is not unreasonable given the relatively small transport distance of NH_3) and (ii) that forests and woodlands cover approximately 10% of the UK land area (derived from the Centre for Ecology and Hydrology (CEH, 2000) Land Cover Map (Fuller *et al*., 2002), where total UK land area is approximately 250,000 km^2). On this basis, the social cost which would arise from reduced carbon sequestration in forests and woodlands across the UK in 2020 following Scenario 1, 2 and 3 reductions in NH_3 emissions can be estimated (Table 3).

N_2O is a greenhouse gas with a global warming potential 310 times that of CO_2 (Houghton and Keller, 2001; DECC, 2009). Increased NH_3 deposition generally causes higher rates of N_2O emission from soil, and the effect becomes more pronounced as deposition rates increase (Skiba *et al*., 1998). Release of N_2O from soils in riparian buffer zones is typically 1% of N_{dep} nationally, but could be anywhere between 6 and 25% of deposition at some locations (Skiba *et al*., 1999; Horvath *et al*., 2003). To provide an approximate estimate of the social value of the climate regulation benefits that could result from reduced N_2O emissions under Scenarios 1, 2 and 3, we assumed that N_2O release represented 5% of N_{dep}, i.e. 50 kg of N_2O emissions per tonne of N deposited.

Hence, allowing for its global warming potential, the effect of a 1 t reduction in N_{dep} on N_2O emissions would be equivalent to a 15.5 t reduction in CO_2e emissions. Valuing this change with DECC's recommended £60 per tonne CO_2e produces a social benefit from reducing N_2O emissions of £930 per tonne reduction in NH_3 emissions or £660 per tonne in 2010 present value. If we further assume that a 1 t reduction in NH_3 emissions produces a corresponding 1 t reduction in N_{dep} across the UK in 2020, then the social benefit which arises in 2020 from reductions in N_2O emissions following NH_3 abatement under Scenarios 1, 2 and 3 can be calculated (Table 3). In reality, the national situation will be more difficult to quantify because the response of N_2O emissions to decreased N deposition is likely to differ between habitats. Full spatial summation of the areas of each habitat type affected would also be required to provide a more detailed assessment.

Identifying the net social benefit of reductions in NH_3 emissions on climate regulation is a major research challenge (e.g. Sutton *et al*., 2008) because the net outcome of changes in CO_2 and N_2O fluxes which would arise across different habitat types is difficult to quantify. In addition, more subtle effects such as suppression of CH_4 oxidation in grasslands, forests and arable systems by N_{dep} (Hutsch *et al*., 1993) have not even been considered here. Social valuation of the changes in climate regulation following implementation of NH_3 control measures therefore also appears likely to present a considerable challenge for some time to come, even though in this case the 'broken links' occur only in the earlier 'scientific' stages of the impact pathway.

3.4. Indicative calculation of benefits to human health

Valuations of the direct effects of air pollution on human health have driven air quality policy historically and thus provide a relevant yardstick against which the valuations of changes in ecosystem service delivery can be compared. Combined research by Stedman (2008) and Watkiss (2008) estimated the economic value of improvements in human health in the UK under different NH_3 emission control scenarios. The basis of their approach is summarised here. Source–receptor models simulate the change in secondary inorganic aerosol (SIA) concentrations in each 1 km grid square across the country in response to a 94 kt reduction in UK NH_3 emissions, enacted as a one-off 'pulse' reduction in 2005 (Stedman, 2008). It was assumed that SIA concentrations affect health outcomes through their contribution to the atmospheric mass concentrations of fine particles ($PM_{2.5}$). The relationship between $PM_{2.5}$ concentrations and health outcomes was assumed to be linear with no threshold. The resident population in each grid square was used to calculate a change in the population-weighted annual mean $PM_{2.5}$ concentration, and hence, using a simple exposure–response relationship together with estimates of disease prevalence, to calculate the health impacts across the country.

On this basis, Watkiss (2008) estimated that health benefits of £1407 would accrue per tonne reduction in NH_3 emissions, primarily through reduced mortality. Watkiss (2008) obtained this estimate by valuing the modelled decreases in mortality and morbidity using the value of life years lost (VOLY) approach (Defra, 2006). UK air quality policy currently uses a VOLY value of £29 000 (in 2005 prices) based on Chilton *et al*. (2004) (Defra, 2006). Watkiss' valuation recognised that the health benefits of a one-off 'pulse' reduction in emissions in 2005 would not be felt until several decades in the future since respiratory and cardiac illnesses primarily affect the elderly, and that the valuation produced therefore had to account appropriately for population demographics over a 100 year time window, starting from the base year of 2005 (Watkiss, 2008). Scaling the results of Stedman and Watkiss' assumed 94 kt emissions reduction in direct proportion with the emissions reductions in our Scenarios 1, 2 and 3 (47, 67 and 87 kt, respectively), and compounding the results forwards from Stedman and Watkiss' 2005 base year to our 2010 base year, produces the estimates in Table 3 of the economic benefits which emissions reduction Scenarios 1, 2 and 3 would deliver nationally for human health. These results are appropriate for direct comparison with our estimates of changes in the value of the climate regulation service and the costs of implementing the different emissions reduction scenarios.

3.5. Comparison of costs and benefits

Table 3 reports the costs for implementing Scenario 1, 2 and 3 abatement (based on Misselbrook, 2007) in which the different mitigation measures are implemented in decreasing order of emissions reduction benefit per unit cost, up to the total estimated emissions reduction capability for the measure concerned. Webb *et al*. (2006) provide further details of the specific reduction measures and the costing mechanism used. Table 3 shows these one-off costs together with the estimated social benefit that would arise annually from following Scenario 1, 2 and 3 abatement via changes in carbon sequestration, N_2O emissions and human health. These results indicate that the costs and benefits which the Scenarios would bring *annually* for the different components of climate regulation are of broadly the same order as the estimated *annual* benefits to human health, and also of roughly the same order as the *total* estimated costs of implementing the abatement measures. Climate regulation is only one of the ecosystem services affected by NH_3 policies. This suggests that when ecosystem service valuations are excluded from cost benefit analysis, as at present, it is likely that substantial elements of social cost and/or benefit are being overlooked when making policy decisions.

4. DISCUSSION

The preceding sections have described some of the challenges encountered currently in attempting to conduct an ecosystem services-based assessment of alternative policies for agricultural NH_3 management. These challenges are clearly considerable, and any ecosystem services-based valuation based on current knowledge must remain incomplete. This section discusses issues which should be considered when contrasting the merits of an ecosystem services-based assessment of policy alternatives with the current practice of basing cost-benefit analysis of alternative policies solely on explicit valuation of the direct effects of emissions on human health and building fabric, whilst relying on a separate, non-monetised critical loads and/or critical levels mechanism to quantify the severity and extent of ecosystem effects.

4.1. Opportunities for holistic appraisal

Table 3 shows that the same abatement policy can deliver both costs and benefits to society by inducing ecosystem effects with opposite consequences for the delivery of particular ecosystem services; our example here is climate regulation. It is clear from Table 1 that further examples of opposing outcomes would emerge from a more complete valuation. For example, N_{dep} may increase production and carbon sequestration from an N-limited ecosystem whilst simultaneously eliminating sensitive species through competitive exclusion (Bobbink *et al*., 2010). A major advantage of an ecosystem services-based assessment is that it aims to provide a systematic assessment of all the ecological effects of policy alternatives, with the intention that information on specific species, processes and habitats should be synthesised into a holistic appraisal. By doing so, an ecosystem services-based assessment can address policy implications for the delivery of important regulating services such water purification, climate regulation and erosion regulation which have often been overlooked hitherto.

Whilst it appears that full monetary valuation of all effects on ecosystem service delivery is not likely to be possible for some time to come, the qualitative stage of an ecosystem services-based policy appraisal can still provide a helpful framework for adaptive management by identifying the implications which alternative emission control policies carry for delivery of different ecosystem services and for possible tradeoffs within, and between, service deliveries. For a holistic accounting of benefits, it is also important to quantify direct effects, other than those on health, which are typically not mediated through impacts on natural ecosystems. For example, important societal benefits of reducing NH_3 emissions may arise from improved atmospheric visibility, while, at a global scale, NH_3 emissions also contribute to the cooling effect of ammonium sulphate aerosol in the atmosphere that counteracts global warming (Ramanathan and Feng, 2008). We are not aware of any attempt to value these effects in the appraisal of policies to regulate emissions, although this may be theoretically possible e.g. by assessing effects of hazy days on tourism and relating the cooling effect of aerosols to carbon sequestration.

4.2. Knowledge gaps identified by the weakest link analysis

Despite the potential advantages that ecosystem services-based appraisal offers, actual implementation is currently frustrated by lack of detailed knowledge. This applies to both the scientific dose–response relationships which link pollutant emissions to predicted ecosystem effects, and the economic functions required to value the predicted changes in ecosystem service delivery. Our assessment (Table 2) identified two main locations for the weakest links in the impact pathways connecting changes in NH_3 emissions with changes in the social value of ecosystem service delivery.

(a) Inadequate or insufficient scientific data to apply to dose–response relationships and/or lack of the relevant dose–response relationships themselves, preventing accurate prediction of physical changes in service flows.
(b) An inability to produce economic valuations of physical changes in service flows, either because available valuation methods do not respond to the measures of ecosystem state and condition which appear as outputs from the scientific stages of the impact pathways, or because the value of marginal changes in ecosystem service flows is highly context-specific, which makes accurate valuation via benefits transfer particularly difficult.

Regarding scientific data, there is a need for information specific to the drivers of impacts on ecosystem services of sufficient quality, and at adequate resolution, to provide the scientific basis for predictions of marginal changes in ecosystem service provision. In terms of biodiversity, there is a lack of knowledge of the effects of species composition change on ecosystem services, and further research is required to increase understanding of the importance of direct effects of N_{dep} on ecosystem function relative to those caused indirectly by changes in species composition. In terms of climate regulation, the net effect of changes in NH_3 emissions on climate regulation needs to be better quantified, accounting for the trade-offs between changes in CO_2, N_2O and CH_4 fluxes from different habitats, interactions between NH_3 emissions and secondary aerosol formation, and carbon sequestration in vegetation and soils. Furthermore, the considerable uncertainty over the timescale of response in different ecosystems will have a significant influence on economic valuation through discounting. Sensitivity analyses will be required to evaluate how different assumptions about timescale affect the predicted changes in service flows and their valuation.

Estimating economic values for changes in ecosystem service flows is also challenging for a number of reasons. Firstly, Table 2 highlights several instances in which economic valuation functions are unable to accept as inputs the ecosystem state or condition variables predicted by the scientific dose–response relationships from earlier links in the impact pathway. This prevents the 'scientific' and 'valuation' sections of the impact pathways from linking together appropriately. For example, a function which predicts the social value of a day's recreation in an upland National Park is unlikely to respond directly to changes in the species diversity of the grassland community in that Park, which might be the typical outcome of a scientific dose–response relationship initiated by changes in NH_3 emissions. A key priority is therefore to re-analyse or re-assess existing scientific data to identify response variables which can link more effectively to ecosystem service delivery and also with the economic functions which value predicted changes in that service delivery. This challenge could be approached from either (or both) direction(s) by:

- adjusting how changes in ecosystem condition or service flows are presented to the public in valuation studies, and/or
- revising or re-configuring scientific models of the effects of emissions reduction so that they produce outputs which describe ecosystem condition and levels of service delivery in ways which will allow the general public to place a more meaningful valuation on the benefits of further emissions reduction.

Economic quantification becomes particularly challenging when site-specific valuations of marginal changes in service delivery are required. This is typically the case for the genetic, cultural, heritage and recreational services associated with biodiversity change and species restoration because of their extreme sensitivity to context (Bateman *et al*., 2006; Defra, 2007a). Given the budget restrictions which inevitably accompany policy appraisal, benefits transfer will inevitably play a prominent role in establishing these site-specific valuations. Benefits transfer is a contested methodology in itself (Desvousges *et al*., 1992; Kristofferson & Navrud, 2005) and it becomes particularly challenging when used to value changes in the delivery of ecosystem services which are highly sensitive to context (Baskaran *et al*., 2010).

Ideally, valuation functions from the literature will contain sufficient variables to control for differences in context. This may be possible where valuations respond to changing spatial context in ways which can be quantified through spatial analysis, e.g. where recreation value is sensitive to road transport distances from major population centres (Bateman *et al*., 2006). However, it is unlikely that context-specific drivers of cultural or heritage value, such as relative level of rarity or specific cultural resonance, can be quantified and adjusted for so easily.

Furthermore, meta-analysis will be unable to control for changes in context when estimating the value of marginal changes in these highly context-specific service flows unless a sufficient number of studies in the literature have reported the levels of relevant context-specific drivers which pertained at their study sites when valuation was undertaken. For example, unless sufficient numbers of valuation studies of changes in species composition in the literature actually report the level of 'rarity' pertaining at their study site then it will not be possible to use meta-analysis to estimate the relationship between the level of rarity and the social value lost through local extinction, or gained through local restoration, of endemics. Studies such as Christie *et al*. (2006), which investigate factors which affect public preferences and willingness to pay for species conservation and restoration at a more general level, could thus prove very useful for identifying when it might be appropriate to attempt benefits transfer-based valuations for changes in delivery of biodiversity-driven ecosystem services.

4.3. Uncertainties in the valuation of changes in health outcomes and in ecosystem service delivery

Table 3 showed that our preliminary estimates of the social value of changes in climate regulation were comparable in scale to the direct human health benefits which are widely used for policy evaluation. However, it is important to emphasise that these estimates of health benefits are also by no means free from uncertainty. The links between NH_3 emissions and aerosol concentrations, between aerosol concentrations and health risks, and the use of residential outdoor concentrations to represent actual exposure all entail considerable uncertainty.

Until relatively recently, a lower reliability rating would also have been assigned to the economic estimate of the value of a life year lost (VOLY). VOLY values were for some considerable time derived primarily from US-based research (Alberini *et al*., 2006; Krupnick *et al*., 2002) and an appropriate, quantified UK or EU-specific VOLY has only emerged more recently following very substantial research effort and expenditure (Johannesson and Johansson, 1997; Soguel and van Griethuysen, 2000; NewExt, 2003; Chilton *et al*., 2004). These research expenditures appear justified given that the VOLY acts as a direct 'per unit' economic multiplier on the mortality endpoints produced by scientific predictions of human health outcomes, and that the valuations which result exert very considerable influence over policy selection.

The DECC-endorsed valuation of 'per unit' changes in greenhouse gas emissions exerts equivalent influence over valuation of the CO_2e endpoints of impact pathways which link air quality policies to climate regulation. Climate regulation outcomes are assuming ever-increasing prominence in policy appraisal. It would, therefore, seem appropriate to direct adequate research effort and expenditure towards establishing quantified, well-bounded estimates of the minimum marginal cost of carbon abatement consistent with achieving UK and global carbon reduction commitments, since this provides the basis for DECC's valuation of changes in CO_2e emissions (DECC, 2009). In common with environment ministries elsewhere, DECC's current carbon valuations draw very heavily on one consultancy study (DECC, 2009; McKinsey and Company, 2009). DECC have undertaken to provide major revisions of their carbon abatement cost predictions every 5 years from 2011, and to provide minor revisions more frequently, but the research basis for these revisions is not specified (DECC, 2009).

4.4. Improvements to assist wider implementation of ecosystem services-based assessments in air quality management

The preceding sections have focussed on our NH_3 case study, but our findings are applicable more widely. In particular, they suggest that more comprehensive implementation of ecosystem services-based assessments for air quality management could be facilitated by improved quantification of the benefits of reduced emissions to the atmosphere. Key issues here include improved quantification of changes in carbon sequestration and fluxes of other greenhouse gases; the complex linkages between atmospheric emissions and water quality and regulation; the time-scale of response of ecosystem services to changes in atmospheric emissions and deposition; the relative importance of direct effects on ecosystem function compared to those caused indirectly by changes in species composition; and improved valuation of marginal changes related to reduced atmospheric emissions.

In general, there is also a need for improved horizontal connectivity along impact pathways, since economic valuation functions are not often configured to respond to the ecosystem state or condition variables which are predicted by currently available dose–response relationships. Improved connectivity could be encouraged by undertaking a number of policy-specific impact pathway valuations for key ecosystem services. Climate regulation would be an excellent choice here.

5. CONCLUSIONS

Our analysis suggests that ecosystem service-based valuation offers considerable strengths and advantages to support a more holistic approach to air quality management. In particular, it provides a framework for systematic assessment of ecological effects, capturing impacts of air quality policy on important ecosystem services and highlighting the potential for pollution swapping. It can be adapted to consider effects at various spatial scales and at a national level. It can also support the definition and application of environmental limits by specifying the particular levels of ecosystem service delivery which these limits protect. However, the full advantages of this approach cannot be realised until there have been substantial improvements in data collection and in the application of methodologies for scientific and economic quantification of the changes in service flows which result from different emissions control policies. Improvements of this type would allow more complete and credible implementation of ecosystem services-based assessment as a tool for air quality management.

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