Towards an Ecosystems Approach for Ammonia– Embedding an Ecosystem Services Framework into Air Quality Policy for Agricultural Ammonia Emissions (Defra NR0120)

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Executive Summary

This study considered whether an Ecosystems Approach can potentially be applied to air quality policy for agricultural ammonia emissions in the UK. The work focused on assessing the feasibility of developing an ecosystem services framework for ammonia emissions using the Defra's introductory guide to valuing ecosystem services (Defra, 2007a). Potential barriers to implementation of the framework were identified and discussed in consultation with key stakeholders.

Approach

An Ecosystem Approach, based on the work of the Millennium Ecosystem Assessment (MEA, 2005) and Convention on Biological Diversity (CBD, 2004), may offer advantages over existing policy and regulatory frameworks. These advantages include:- a holistic approach considering the whole range of effects of a particular policy measure as a starting point; the linkages that can be made between ecosystems services and constituents of human well-being such as health and basic materials for a good life; identification of synergies, antagonistic effects and tradeoffs; inclusion of regulatory services which are under-represented in current UK policy; and insight into the full costs and benefits to society of policy measures.

An environmental baseline was first established to assess the effects of current ammonia emissions on ecosystem services. This initial assessment highlighted significant potential links between ammonia emissions and positive and negative impacts on a large range of ecosystem services.

Marginal changes to ecosystem services that could result from different ammonia reduction scenarios for 2020 were then assessed using a qualitative (i.e. change from baseline) method. The methodology used was developed to be as compatible as possible with existing information and regulatory structures in the UK. The analysis considered the likelihood of Biodiversity Action Plan (BAP) broad habitat types being affected by reduction of a specific ammonia source according to their spatial distribution. Effects from ammonia sources were classified as those arising from point sources (e.g. intensive pig and poultry housing); diffuse sources (e.g. fertilizers and grazing); and intermittent sources (e.g. slurry applications). The effects of ammonia reduction scenarios were then assessed according to their relative impacts on these different sources.

The marginal changes identified were then used as the basis for determining which ecosystem service effects should be considered in the quantitative stage of the analysis. The linkage of these marginal changes in ecosystem services to human welfare was then considered and an assessment made of the potential to make an economic valuation of these changes and conduct a cost-benefit analysis. At present this is a partial exercise since comprehensive data on all ecological responses and valuation of associated services are lacking.

Main Results

- The most important, and relatively well understood, positive changes to ecosystem services resulting from reduced ammonia emissions were related to air and water quality, species composition and climate regulation (i.e. decrease in greenhouse gas (GHG) emissions from soils). Important negative changes occurred where the fertilizing effect of nitrogen deposition had had a beneficial effect on harvested goods and carbon sequestration by vegetation and where changes in methods of slurry storage and application may lead to decreased ammonia emission at the expense of increased GHG emissions or nitrate leaching from soils (i.e. pollution swapping).
- Wetlands, broadleaf woodlands and grasslands were predicted to show the greatest change in ecosystem services due to their large geographical overlap with ammonia sources in the UK.
- Preliminary calculations, for changes in climate regulation only, show that the scale of benefits and costs is likely to be of a similar order of magnitude to that estimated for the human health impacts of ammonia. The analysis also suggests that the costs of the more stringent ammonia emission reduction being considered for 2020 are unlikely to be matched by savings from health benefits alone and that including benefits related to ecosystems services could potentially alter this balance, especially given the magnitude of the figures estimated and the fact that many of the effects on ecosystem services are positive (providing societal benefits).

Comparison of ecosystem approach with existing air quality management

The strengths that this study has identified for the application of the Ecosystems Approach to air quality management for ammonia can be related to Defra's core principles (Defra, 2007b) in the following way:

- It provides a framework for systematic assessment of ecological effects and allows information on specific species, processes and habitats to be synthesised;
- It captures important regulating effects and can highlight pollution swapping potential, such as those for water quality and climate regulation, which are not currently considered in the National Air Quality Strategy;
- It can be adapted to include consideration of effects at various scales and at a national level;
- It can support the definition and application of environmental limits by defining the specific ecosystem services which they can protect;

- It provides a framework for a comprehensive holistic assessment, involving human health, ecosystem services and the built environment;
- It provides a framework for adaptive management, since the implications for different ecosystem services of alternative emission control policies, and the possible tradeoffs involved, can be clearly identified.

The comparison with existing air quality management also showed that the use of the ecosystem services framework would clearly allow more explicit consideration of the consequences of critical load and level exceedance, and hence would add a new dimension to policy evaluation. The weaknesses of the Ecosystems Approach that were identified mainly relate to its generic application, rather than to the specific context of air quality management.

Consideration of the barriers to implementation

This assessment considered specific (ammonia related) and general barriers to implementing an Ecosystems Approach and it also benefited from comments on an interim report from a small group of air quality management experts and stakeholders. The following are the salient points:

- Research is needed in certain key areas to overcome limitations of available knowledge on the benefits of reduced ammonia emission. These include:- quantification of carbon sequestration in vegetation and soils; net nitrous oxide emissions from agricultural land and other habitats; the complex linkages between ammonia emissions and water quality and regulation; the relative importance of direct effects of nitrogen deposition on ecosystem function compared to those caused indirectly by changes in species composition; and improved valuation of marginal changes related to reduced ammonia emissions.
- Only economic valuation functions produced by the most modern methodologies will be appropriate to accurately value the socio-economic benefits of implementing ammonia control; the vast majority of existing studies in valuation provide an unsuitable basis for benefits transfer-derived estimation of the value delivered by ammonia control measures.
- There is a need to communicate the ecosystem services concept in a non-technical way suitable for the decision making process - clarifying links between ecosystem services and the positive and negative outcomes for society using the ‘ecosystems cascade approach’ is one way that this could be achieved.
- Our analysis shows that application of the Ecosystems Approach can highlight conflicts between the priorities of existing policy areas, such as conservation of plant species, maintenance of water and air quality and issues of climate change regulation, however, the extent to which the Ecosystems Approach could be used to resolve such conflicts is very uncertain.
- The large uncertainties involved in applying the Ecosystems Approach may mean that policy instruments may require a precautionary tone, which may be problematic for certain stakeholders.

More generic challenges that implementation of the Ecosystems Approach would have to overcome include: obtaining public understanding and acceptance, development of the skills and inter-disciplinary knowledge base that is required and adjustments that would need to be made to regulatory frameworks for the management of multiple ecosystem services.

Recommendations

- Development is needed of methods for dealing with the high degree of complexity introduced by application of the Ecosystems Approach.
- If detailed application of the Ecosystems Approach is limited to a few key ecosystem services, methods of identifying objectively the key areas need detailed development and appropriate stakeholder involvement.
- The lack of available data quantifying the value of marginal changes in the delivery of individual ecosystem services is a major barrier which can only be overcome by investment in new data collection; analysis to identify the key data gaps that prevent reliable valuation for a particular issue is possible using a ‘weakest link’ approach.
- This study clearly reveals many uncertainties and inadequacies concerning the information available for implementing an Ecosystems Approach for ammonia, but a strong conclusion from this study is that the qualitative stage of the impact pathway approach is already a very useful tool to aid integrated policy development.
- Conflicts between different policy objectives can be made transparent by application of the Ecosystems Approach, but, especially given the current limitations in valuation and the potential for low national valuation of services that are highly valued by certain groups, a framework is needed for how the Ecosystems Approach could be used for resolving these conflicts.
- Case studies with stakeholder involvement in areas close to sensitive habitats would provide demonstrable examples of the ecosystems based approach and could also allow links to other policy drivers to be made more explicit.

1. Introduction

1.1 Background

Emissions of ammonia (NH_3) to the atmosphere can have significant effects on a range of sensitive ecosystems and on human health (i.e. through the formation of secondary inorganic aerosol). Its ecological effects relate to both increased deposition of nitrogen and acidification. As emissions of sulphur dioxide and nitrogen oxides have fallen faster than those of ammonia in recent years, the role of ammonia as a pollutant has become more significant. In addition, there is some evidence that the ecological effects of reduced nitrogen are greater than those of oxidised nitrogen.

In the UK, efforts to date have been placed on:- defining critical loads and critical levels above which adverse effects of nitrogen deposition or gaseous ammonia can occur; modelling ammonia transport, deposition and exceedance of critical loads and levels; quantifying ammonia emissions; and assessing the costs and benefits of different abatement strategies at the national scale. 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) commit the UK to a national target for 2010 of 297 kt NH_3 year^{-1}. The CLRTAP Gothenburg Protocol and EU NECD targets are likely to be revised in the next few years to set new, lower emissions ceilings to be met by 2020. These are likely to be challenging for the UK

Defra's emission projections indicate, that although the UK is likely to meet its NECD emission ceiling target for ammonia in 2010, this reduction in emissions will be insufficient to protect most UK ecosystems from the effects of atmospheric ammonia. For example, recent estimates of critical load exceedance in the UK (Hall *et al.*, 2004) show that about 65% of UK ecosystems (by area) are estimated to receive atmospheric nitrogen deposition that is in excess of the critical load for adverse ecological effects, and that substantial exceedance will remain in 2010. Since reduced nitrogen (the sum of ammonia and ammonium, NH_x) makes a major contribution to total nitrogen deposition, ecological changes due to ammonia are expected in sensitive ecosystems over much of the country.

However, the Air Quality Strategy for England, Scotland, Wales and Northern Ireland (Defra, 2007c) states that it does not consider it an appropriate time to set and apply concentration-based objectives for ammonia based on exceedance of critical ecological thresholds, without more detailed consideration of wider environmental issues and regulatory frameworks. This is because of the complex issues related to actual policies for reduction of ammonia emissions from agriculture, which have a number of environmental implications related to other policy drivers such as the Nitrates Directive and Water Framework Directive. Rather, it states that a holistic and strategic approach needs to be taken to tackling nitrogen emissions through nutrient management in a wider context of losses to water and air and impacts on climate change.

Such an integrated approach is consistent with the UK government's commitment to embed an Ecosystems Approach (Defra, 2007b) in existing policy/ regulatory frameworks, so that they provide a better focus on human well-being and on providing value for money. By comparison with the existing focus on critical thresholds, an Ecosystems Approach, based on the work of the Millennium Ecosystem Assessment (MEA, 2005) and the Convention for Biological Diversity (CBD, 2004) may offer important advantages, such as:-

- (i) a holistic assessment that considers the whole range of ecosystem services as a starting point;
- (ii) inclusion of regulating services, such as ecosystem controls on fluxes of pollutants in land-water-air systems, which are currently under-represented in UK policy;
- (iii) identification of negative externalities, ancillary benefits and trade-offs of policy measures;
- (iv) insight into the full costs and benefits of policy measures.

However, to date, there has been no systematic evaluation of whether the Ecosystems Approach provides a practical alternative to current approaches to air quality management. The broad aim of this project, therefore, was to determine if an Ecosystems Approach is currently technically feasible, and desirable, for the holistic assessment of impacts of air pollution on the environment. The analysis presented in this report uses the case of ammonia emissions abatement to illustrate the principles of the application of the Ecosystems Approach. An assessment based on the Ecosystems Approach is potentially relevant for all air pollution issues, and could be used more widely in future policy analysis if its application to ammonia appears to be practicable.

1.2 Objectives

The specific technical objectives were:

1. To test the potential for an Ecosystems Approach to make a meaningful holistic assessment of the impacts on ecosystem services of different abatement options for ammonia emissions to the air at national level in the UK;

2. To compare the Ecosystems Approach with current conventional air quality management approaches to determine the relative strengths and weaknesses of each;

3. To identify potential barriers, and their causes, to the implementation of an Ecosystems Approach for air quality policy development at national level in the UK;

4. To consider the type of short and long-term changes to the policy /regulatory framework, and research and development needs, that may be needed to address barriers identified in 3.

1.3 Outline of report

This final project report is based on the interim report entitled 'Applying the Ecosystems Approach to air quality policy (Defra NR0120)' submitted to Defra in July 2008 and comments subsequently received from the Defra project committee.

The interim report was also sent to some key stakeholders and experts regarding ammonia air quality policy in the UK, and replies were received from: Tom Tew (Natural England), Simon Bareham (Countryside Council for Wales and Joint Nature Conservation Committee (JNCC)), Mark Broadmeadow (Forestry Commission England), Tom Misselbrook (Institute of Grassland and Environmental Research (IGER)), and Ken Smith (ADAS). The stakeholders were asked the following questions about the interim report:

i. How would the implementation of an Ecosystems Approach in the UK fit with your organization's objectives and priorities?

ii. Do you agree with the general approach taken in our report (see Section 2)?

iii. How do you see the comparative strengths and weaknesses of the Ecosystems Approach with existing approaches (see Section 3)?

iv. What do you feel are the main barriers to the implementation of the Ecosystems Approach in the UK (see Section 4 and 5)?

We are very grateful for their comments which are used anonymously in this final report. **It is important to note that the stakeholder comments should be viewed as personal comments from within an organisation and not as official organisational stakeholder responses.**

The work undertaken to meet objectives 1-4 is described in turn in Sections 2-5 of this report. The main conclusions, future research priorities, and communication of the findings are then described in Sections 6-8. The body of the report only summarises the main features of the analysis associated with Objective 2; most of the details of this analysis are contained in a separate Technical Annexe to the report.

2. Developing an ecosystem services framework for ammonia (Objective 1)

2.1 Approach and methodology development

This report focuses on developing an ecosystem services framework for agricultural ammonia emissions, following the Defra guidelines (Defra, 2007a), and then considers the feasibility of implementing it as part of an Ecosystems Approach, including full costing of changes to ecosystem services, human health and the built environment.

The Ecosystems Approach (Defra, 2007b) provides a compelling framework for policy makers and managers for the management of ecological-social systems. It is a holistic approach that allows, at least in principle, evaluation of the full costs and benefits of a particular policy option. Having these full costs and benefits on the table allows the relative merits of competing policies to be assessed.

Central to the Ecosystems Approach is the concept that natural ecosystems provide services for people (e.g. flood control, crop pollination, water purification, recreation and food stuffs) and that by knowing how people perceive the relative worth of these different services, consensus can be reached on how best to manage those services for the maximum benefit to society. Management policies to ensure the delivery of preferred services from an area may include the control of environmentally damaging activities, such as pollution, here specifically ammonia emissions. These emissions are known to impact on natural ecosystems and are thus likely to affect the service provision from ecosystems. Within the work of the Millennium Ecosystem Assessment (MEA, 2005) and Convention on Biological Diversity (CBD, 2004), the Defra's Ecosystem Approach is based on, the impacts on ecosystem services can then be linked to constituents of human well-being such as health and basic materials for a good life.

In addition, and quite distinct from the ecosystem services dimension, are the direct effects of air-borne pollutants on human health and on physical structures such as buildings. These effects are extremely important to quantify as part of the full costing of a policy in the Ecosystems Approach, but they are not mediated through their impacts on natural ecosystem services. However, for the effects of air-borne pollutants on human health there is an ecosystem service dimension as ammonia and related compounds in the atmosphere can be removed by the filtering effect of vegetation and soil nitrogen cycling can be affected by farm practices, such as use of manures, and lead to altered ammonia emissions to the atmosphere. The Ecosystems Approach will take these factors into account but in an attempt to make the methodology developed in this study as compatible as possible with existing information and regulatory structures in the UK, the main valuation of human health effects of ammonia emissions is considered in terms of changes to the national emission of ammonia as described in Section 2.6.

Our analysis of the environmental implications and the potential for valuing the marginal effects on ecosystem services of different emission control scenarios for ammonia proceeded in five stages, based on the recommendations of Defra (2007a) for the application of the Ecosystems Approach, as follows:-

1. Establish the environmental baseline;
2. Identify and provide qualitative assessment of the potential impacts of policy options on ecosystem services;
3. Quantify the impacts of policy options on specific ecosystem services;
4. Assess the impact on human welfare;
5. Estimate the economic value of changes in ecosystem services.

The scenarios adopted for analysis are based on the work of Misselbrook (AQ0602 Underpinning evidence for the UK ammonia strategy, 2007). This identifies the most cost-effective combination of measures needed to meet different national emission ceiling targets for 2020. We use all three emission reduction scenarios within the work of Misselbrook (2007), relating these to the stages of analysis above as follows.

For the qualitative analysis we use Scenario 1 (250 kt target) identified by Misselbrook (2007), as it includes the four different types of policy measures that Defra are considering, namely, those connected with fertilisers, land spreading, pig and poultry housing and slurry storage. The results of the qualitative analysis are then used to consider the effects of applying Scenarios 2 and 3 as they entail more stringent application of requirements to reduce emissions from two of the four activities analysed under Scenario 1, i.e. storage (230 kt target under Scenario 2) and housing (210 kt target under Scenario 3).

The Defra guidelines (Defra, 2007a) for establishing the environmental baseline and for the qualitative analysis of the potential impact of policy options have been adapted for this study to reflect the complex nature of ammonia emissions and deposition and its impacts in the environment. Firstly, we re-group the policy measures contained in the scenarios into three generic types, based on the types of environmental impact that they may cause (see also Annexe II). These are as follows:-

Point source impacts – these impacts relate to activities which release large amounts of ammonia originating from a clearly defined source (e.g. intensive pig and poultry housing) within a relatively small geographic area, and are primarily caused by gaseous ammonia.

Intermittent impacts – these impacts relate to release of large amounts of ammonia from intermittent sources within a limited period of time (e.g. occasional peaks in ammonia concentrations from intensive slurry application in early spring).

Diffuse impacts – these impacts relate to the contribution of large areas of relatively low ammonia emissions (e.g. fertiliser application over large areas of arable farming, grazing emissions) to regional total atmospheric nitrogen deposition.

Point source and intermittent impacts are often more immediate and apparent (e.g. odour, visible damage to trees) whilst diffuse effects occur over longer time-scales with broader, less visible impacts (e.g. on soil formation, climate regulation etc.). It is recognised that the links between sources, policy measures and impacts described above are not entirely mutually exclusive. For example, ammonia released from point and intermittent sources, such as large pig and poultry units and slurry application will contribute to regional background nitrogen deposition as well as to locally elevated concentrations. Similarly, there are interactions between these ammonia sources and impact types, for example, as point sources may to some extent cause intermittent impacts, dependent on fluctuations in wind direction and other meteorological factors.

Nevertheless, this division of policy measures allows a more differentiated application of the Ecosystems Approach. This is because different ecosystem processes and services may be affected in different ways by these three types of impact. To take one simple example, it has been argued that carbon storage in forest ecosystems across Europe is positively related to nitrogen deposition (Magnani et al., 2007) and hence could benefit from current levels of ammonia emissions. However, there is also evidence of adverse effects of ammonia at high concentrations on tree vitality and growth close to intensive livestock units, and hence a possible adverse local effect on carbon storage.

Secondly, the environmental baseline analysis, and that of the impacts of policy scenarios, for ecosystem processes and services, has to recognise that the environmental consequences of ammonia emissions differ in different types of ecosystem. We therefore have used the Biodiversity Action Plan (BAP) Broad Habitats as a framework for our analysis. Table 2.1 lists the major effects of NH_3 emissions in these different habitats based on our expert judgement. While some ecosystem services result from processes in a mosaic of ecosystems, a habitat analysis has been used because:-

(a) much of the analysis of the impacts of ammonia in the literature is habitat specific;
(b) it facilitates a comparison with current Air Quality Management using the critical load approach, which is habitat specific.

The broader strengths and weaknesses of a habitat focussed application of the Ecosystems Approach are discussed in more detail by Haines-Young et al. (2007).

The matrix analysis, which is described below, therefore considers environmental baseline impacts, and the changes under different emission control scenarios, in terms of the point source, intermittent and diffuse impacts on a range of broad habitat types. In this analysis of scenarios, the spatial distribution of policy measures in relation to the spatial distribution of the impacted habitats was also considered by assessing the proportion of the habitat located in regions which are dominated by different types of ammonia sources (e.g. slurry spreading, pig and poultry units), based on the analysis of spatial distribution of ammonia emissions by Hellsten et al. (2008).

Assessment of the potential for quantitative analysis of the impacts of abatement measures focused on the availability of the information required and the techniques that would be needed to assess the outcomes of different emission control scenarios. This part of the analysis focussed, in particular, on the quantification of exposure-response relationships between ammonium deposition, or ammonia concentration, and ecological responses, and the potential to express these in terms of specific ecosystem services. We assume that adequate atmospheric transport models are available to relate changes in NH_3 emissions to concentrations and deposition to different locations and habitats.

As well as the impacts on ecosystem processes and services that arise from exposure to gaseous ammonia or from its contribution to total atmospheric nitrogen deposition, the analysis considered the significance of potential ‘pollution swapping’ for different policy measures, including for examples, changes in N_2O emissions or nitrate leaching. These impacts were assigned to the relevant Broad Habitat, for example arable and horticultural land, improved grassland, or rivers and streams.

Table 2.1. UK BAP broad habitat types (UK Biodiversity Partnership, 2007) and the major effects of NH_3 concentrations and deposition. In most terrestrial habitats, N deposition is also associated with N accumulation in soils, and associated increases in organic matter and carbon.

UKBAP habitat	NH_3 effects
Rivers and Streams	• Eutrophication • Acidification
Standing Open Water and Canals	• Eutrophication • Acidification
Arable and horticultural	• Injury to horticultural foliage and fruits from acute exposure
Broadleaved, Mixed and Yew Woodland	• Increased tree biomass • Decreased species numbers • Decreased fungus and lichens • Increase in nitrophilous ground vegetation • Increased soil N cycling and nitrate leaching
Coniferous Woodland	• Increased tree biomass • Decreased fungus and lichens • increased algae • Increase in nitrophilous ground vegetation • Increased soil N cycling and nitrate leaching
Acid Grassland	• Increased biomass • Decreased species numbers • Eutrophication • Acidification and N leaching
Calcareous Grassland	• Increased biomass • Decreased species numbers • Increased N cycling rate and N leaching
Neutral Grassland	• Increased biomass • Decreased species numbers • Eutrophication • Acidification and N leaching
Improved Grassland	• Increased biomass • Acidification from intermittent exposure
Dwarf Shrub Heath	• Transition of heather to grass • Decrease in lichens • possible leaching if heather canopy is reduced • increased decomposition rates and loss of stored carbon
Fen, Marsh and Swamp	• Decreased species numbers • reduction of peat mosses • increased N in water
Bogs	• Decreased species numbers • reduction of peat mosses • increased N in water • Loss of stored carbon
Montane Habitats	• Decreased species numbers • Decrease in peat mosses and lichens • possible leaching if heather canopy is reduced • Increased decomposition rates and loss of stored carbon
Supralittoral Sediment	• Biomass increase • Increase in N leaching

2.2 Establishing the environmental baseline

In line with Defra's wish to develop a more holistic and integrated approach towards policy and decision making, which considers whole ecosystems (i.e. impacts across air, water, land and soil and biodiversity), the environmental baseline was established by relating as many ammonia impacts as possible to ecosystem services to give a comprehensive assessment of the potential effects. Before undertaking a detailed assessment of the baseline impacts of NH_3 emissions on particular habitats, an initial scoping was undertaken to identify the presence of any significant links between ecosystem services and ammonia emissions. As suggested in Defra's guide to valuing ecosystem services (Defra, 2007a), the checklist of ecosystem services used in this study was adapted to suit application to the ammonia problem. The results of this exercise are summarised in Table 2.2.

The results show significant potential links between NH_3 emissions and a large range of ecosystem services. The three exceptions are natural hazard regulation, pest regulation, and disease regulation. While there is evidence that increased concentrations of ammonia can increase the prevalence of insect herbivores and fungal diseases of plants, these were considered to be drivers of changes in plant growth or species composition, and hence inclusion in their own right would result in 'double counting' of the impacts of NH_3 emissions. Similarly, increased plant growth caused by NHx inputs could provide improved conditions for disease bearing pests (e.g. bracken as a host to Lymes Disease). However, these are secondary effects caused by other direct N impacts, such as increasing primary productivity. In the case of water purification, it is not known how increases in NH_3 emissions may affect microbial process responsible for the degradation of pollutants such as pesticides, and assessment of effects on this ecosystem service is also problematic. Therefore, these four ecosystem services were not considered further in this analysis, although, based on some stakeholder comments, they might have to be revisited if the ecosystem service approach was implemented in the UK.

It should be noted that supporting services are included in our qualitative analysis but they are dealt with carefully in the valuation part of the analysis because of the potential for double counting. Supporting services (e.g. nutrient cycling) are a good example of 'intermediate' services that impact on other 'final' services and they will therefore not be valued directly in this study but are included by taking account of their impact on other ecosystem services. Section 2.6 shows how supporting services are the ecological functions that form the link between ecosystem services affected and the positive and negative outcomes for society that can be valued.

Tables A1 and A2, shown in Technical Annex I, show the qualitative assessment for the environmental baseline of the impacts on ecosystem services of ammonia emissions emanating from point sources and diffuse sources respectively. The effects of intermittent emissions, e.g. from land spreading, were not considered in this baseline analysis, but are taken into account later in the analysis when different emission control scenarios are considered. It is important to emphasise that the environmental baseline does not contain any consideration of spatial distribution of habitats or NH_3 sources. Hence Tables A1 and A2 assess the potential effect of ammonia on ecosystem services without evaluating the likelihood of its occurrence. This comprehensive analysis is included for clarity, and to provide the platform for more detailed qualitative analysis of policy implementation, where we use spatial data to evaluate the importance of each measure.

The tabular approach suggested by Defra (2007a) has been modified to include the different habitat and source types (as discussed in Section 2.1 above). There are a number of uncertainties associated with using a habitats approach for assessing the impacts of NH_3 on ecosystem services. Most broad habitat types will undergo changes in species composition and reduction in species richness when subjected to high levels of NH_3 concentrations and deposition. However, considerable research is required into how these habitat-level changes will affect the functional groups present and subsequent ecosystem services. Evaluating the effect of change, for example, in cultural services is highly subjective and is likely to vary greatly depending on region and country (e.g. the cultural value given to bilberry collection in Sweden is less clearly recognized in the UK).

We recognise that there already exists a body of data stating the relative importance of habitats for different ecosystem services, which was produced under Defra contract NR0107. Integrating these assessments was beyond the scope of this analysis, which has been carried out from the perspective of NH_3 pollution specifically. Furthermore, while the assessment under contract NR0107 was validated using stakeholder input in a series of workshops, we have not been able to obtain such validation for the assessments in Tables A1 and A2. For these reasons, the assessment carried out in this report is based on the expert judgement of our team alone. If there was an intention to develop the analysis and its application further, it would benefit from input from expert stakeholder groups and from integration with other more general schemes linking ecosystem services to specific habitats.

Table 2.2 Summary of effects of ammonia emissions on ecosystem services.

Ecosystem Service	Effect of ammonia emissions
1. Provisioning Services	
Ecosystem goods	Production of goods (e.g. food, fuel, fibre) can be increased and decreased.
Water quality	Acidification and eutrophication of surface waters can be caused by direct deposition or by leaching from terrestrial ecosystems.
Biochemical/genetics	Abundance of species can be reduced (or increased in certain circumstances) and community composition can be changed in both terrestrial and aquatic ecosystems.
2. Regulating services	
Air-quality regulation	The growth of trees and tall vegetation can be affected, altering their ability to remove air pollution, while NH_3 emissions contribute to formation of secondary particulates
Climate regulation	Carbon sequestration, methane fluxes and nitrous oxide production are all affected
Water regulation	Effects on peat creation and forest growth can affect water storage and interception.
Water purification	The capacity of wetlands to remove nutrients from water may be reduced by excess atmospheric inputs.
Natural hazard regulation	No significant direct effects
Pest regulation	No significant direct effects
Disease regulation	No significant direct effects
Pollination	Both vegetation composition and flowering intensity can be affected.
Erosion regulation	Increases and decreases in vegetation cover can be caused, leading to changes in rates of erosion
3. Supporting services*	
Soil formation	Detrimental effects can occur on peat formation, but successional change and soil formation can be enhanced in other soils
Primary production	Increase of biomass in N limited terrestrial and aquatic habitats
Nutrient cycling	Rates of soil mineralization can be increased and production of greenhouse gases and nitrate leaching can be enhanced Increased soil N accumulation can occur and may be associated with increased C sequestration
4. Cultural services	
Recreation and tourism	Large changes in terrestrial and aquatic species composition may affect field sports and ecotourism
Aesthetic	Significant if it is assumed that changes from the status quo (e.g. changes in species composition) are negative.
Educational	Reduction in species rich habitats as sites for study
Cultural heritage	Loss of iconic species

Note: Including Supporting Services can lead to double counting

In Technical Annex I, we also summarise in general terms the rationale behind the scores assigned to specific services and habitats. This provides a simple justification of each of these scores, to facilitate further development of the approach, but we emphasise strongly that the focus of our analysis is on broader issues associated with the potential for application of an Ecosystems Approach in the context of ammonia emissions from agriculture.

2.3 Qualitative assessment of the potential impacts of policy options on ecosystem services.

This section describes the effects on ecosystem services of the ammonia emission reduction measures in Scenario 1, described in detail by Misselbrook et al. (2007). This consists of a series of NH_3 abatement strategies intended to reduce total annual UK emissions to below 250 Kt by 2020. These measures are summarised in Table 2.3, which shows the specific measures, the absolute and relative reductions achieved, and the type of emission, using the three classes of emissions identified in Section 2.1.

Table 2.3 also summarises the potential for 'pollution swapping' as a result of these measures, in which reduced NH_3 emissions are 'traded' for other N losses as nitrate or N_2O. The potential may also exist for NH_3 emissions to increase elsewhere in the nutrient cycle – e.g. reduction in emissions at the housing and storage stages can lead to increased N content in slurry and greater NH_3 losses at the spreading stage. Misselbrook et al. (2007) recommend that all measures be used in conjunction to minimise N losses at all stages. However, it should be noted that potential pollution swapping for nitrogen may be acceptable in some circumstances, while being a significant threat in others. For example, measures that simply reduce ammonia emissions and thereby keep more nitrogen in the farming system, in principle, may allow increased losses of other N forms later on – simply because less is lost as NH_3 to the atmosphere. Such pro-rata changes would not be considered as a major threat in well managed situations. By contrast, measures for NH_3 abatement that specifically make the saved nitrogen more liable to other N loss processes would be considered a more significant pollution swapping threat.

These illustrations indicate the need to take a holistic approach to ammonia and nitrogen mitigation. This is based on the simple principle of the nutrient cycle in which any N that would have been lost as NH_3 is retained in the manure/slurry and therefore available for loss in another form. It should be noted however, that whilst the *potential* for nitrate leaching may be augmented due to NH_3 abatement measures, if the extra N that remains in the manure/slurry is actually known and accounted for, this potential leaching need not be realised if manure applications are well matched to plant requirements.

Full details of the qualitative analysis are given in Technical Annex II, including a full description of point, diffuse and intermittent emission types defined in Table 2.3. Where possible, the potential for pollution swapping is included in the analysis.

Table 2.4 provides a summary of those services that are shown by the qualitative analysis to be most affected by Scenario 1 abatement measures. The full method and results are given in Technical Annex II. In general, our analysis demonstrates that most effects of NH_3 abatement result in positive changes to ecosystem services, the exceptions being for those habitats where N fertilisation had a beneficial effect on harvested goods (e.g. woodlands) and those that are subject to direct fertilisation (arable and improved grassland) where changes in slurry storage and application are likely to lead to increases in pollution swapping. The services that are expected to be significantly affected across the broadest range of habitats are water quality regulation and climate regulation (Table 2.4), reflecting the widespread impact of NH_3 on the potential for nitrate leaching and N_2O production in soils. Biochemical/genetic services are also an important beneficiary, based on the assumption that reducing community and species changes will maintain as-yet unused resources.

Table 2.3 Scenario 1 NH_3 abatement measures (Misselbrook et al. 2007) and their emission types.

Abatement measure	Implementation steps	Emission reduction (Kt)	% reduction from baseline*	Emission type	Potential pollution swapping
Land spreading – rapid incorporation – reduced emission application techniques	• Rapid incorporation of all manure and slurry by disc cultivation after application to arable land • Apply slurry to grassland using a trailing shoe • Apply beef cattle slurry to arable land (growing crops) by trailing hose (band spreading)	31.9	14.3	Diffuse, intermittent	Potentially increases risk of N losses via nitrate leaching or N_2O emissions, by retaining more N in the agricultural system.
Slurry storage	• Encourage natural crust formation on cattle slurry tanks and lagoons • Install a floating/flexible cover to pig slurry tanks and lagoons	7.8	3.5	Diffuse	Greater NH_3 emissions after spreading (if not used with rapid incorporation) Increases risk of N losses via nitrate leaching or N_2O emissions
Pig + poultry housing	• Introduce manure drying to layer housing - deep-pit caged and perchery systems • Introduce in-house manure drying for breeding hens, pullets, broilers and turkeys • Improved slatted floor design for fattening pig and sow housing	3.1	1.4	Point-source	Greater potential for N loss during storage and spreading (NH_3, N run-off, nitrate leaching, N_2O emissions)
Fertiliser	• Replacement of urea with ammonium nitrate	16.2	51.7	Diffuse, intermittent	Increases risk of N losses via nitrate leaching or N_2O emissions

* Percentage reduction from baseline calculated as the reduction from current emissions for respective type e.g. livestock or fertiliser

Although such an analysis is beyond the scope of this report, it is likely that this approach would highlight differing priorities for emission control for each country, for reasons summarised below.

England

England is dominated in the south and east by emissions from intensive animal units and there is a high overlap of these areas with fen marsh and swamp habitats. In the north and west, emissions from manure storage and spreading (largely cattle) dominate. England has a mix of most habitats and a more detailed regional analysis may indicate the importance of specific abatement measures in different areas. Background emissions are dominant only in small areas.

Scotland

The highlands of Scotland are dominated by background emissions, with lowland areas largely covered by areas dominated by cattle manure sources. Intensive animal unit sources are only dominant in relatively few 'hotspots', so it is likely that manure storage and spreading measures will be most important in Scotland. Scotland also contains the majority of montane habitat and coniferous woodland in Britain, making these a potential priority; for impacts on these habitats, measures related to diffuse ammonia emissions would therefore be most relevant.

Wales

There is almost no dominance of intensive animal units in Wales, with animal-source emissions dominated by cattle and sheep. It is likely that manure storage and spreading measures will have the greatest effect in this country. This is a similar position to Scotland in terms of sources. However, the fact that background sources are dominant almost nowhere in Wales, and that absolute rates of NHx deposition are higher than in Scotland elevates the importance of these measures further in Wales.

We have not extended our habitat-based analysis to Northern Ireland. However, it is clear from the high NH_3 concentrations and high rates of NHx deposition experienced there that measures to reduce NH_3 emissions could have significant effects on the ecosystem services provided by the dominant habitats in this region. Hence any more detailed spatial analysis in the future should be extended to Northern Ireland.

As in the earlier stages of the analysis, we do not present this as a definitive statement of the most significant impacts on ecosystem services. More detailed discussion with key stakeholders and key experts would be needed to reach a more informed view on which are the most important impacts, especially if this type of analysis is to provide the basis for decisions on which policy measures to adopt in different parts of the country. It might be argued, for example, that Table 2.4 underestimates the importance of effects on carbon sequestration, and overestimates effects on water quality in arable and improved grassland areas. More detailed evaluation of the 'pollution swapping' aspects would also be essential. In a full lifecycle analysis it still needs to be demonstrated that reducing ammonia emissions by low emission spreading techniques leads to an overall increase in N_2O emissions. This is because atmospheric deposition of the ammonia emitted can lead to N_2O emissions in other locations (e.g. for semi-natural ecosystems). Thus abating ammonia emissions in this way reduces the N_2O emissions from semi-natural ecosystems, matched against an increase in N_2O emissions from the agricultural grass/cropland. Table 2.4 identifies this trade-off of services clearly, but the extent to which this trade-off is "N_2O neutral" would require more detailed quantitative analysis and remains a matter for further research.

Table 2.4 Summary of ecosystem services most affected by Scenario 1 NH_3 abatement measures

Service	Habitat	Emission Type	Value (benefit/disbenefit)	Reason
Ecosystem goods	Rivers & streams	Point, diffuse, intermittent	+2 (water quality, fish, recreation)	Decrease in NH_3 input from all sources will increase availability of goods (fisheries)
	Standing open water and canals	Point, diffuse, intermittent	+2 (water quality, fish, recreation)	Decrease in NH_3 input from all sources will increase availability of goods (fisheries)
Water quality	Arable and horticultural	Diffuse, intermittent	-2 (water quality, fish, recreation)	Spreading measures will decrease NH_3 emissions but could increase N leaching
	Improved grassland	Diffuse, intermittent		
	Broadleaved mixed and yew woodland	Point, diffuse, intermittent	+2 (water quality, fish, recreation)	Lower NH_3 deposition will decrease N leaching
	Coniferous woodland	Diffuse, intermittent		
	Acid grassland	Diffuse		
	Calcareous grassland	Diffuse, intermittent		
	Neutral grassland	Diffuse		
	Dwarf shrub heath	Diffuse		
	Fen marsh and swamp	Point, diffuse, intermittent		
Biochemical/genetics	Rivers & streams	Point, diffuse, intermittent	+2 (valuable characteristics of species preserved)	Lower NH_3 deposition will reduce species loss
	Standing open water & canals	Point, diffuse, intermittent		
	Broadleaved mixed and yew woodland	Point, intermittent		
	Coniferous woodland	Intermiittent		
	Calcareous grassland	Intermittent		
	Dwarf shrub heath	Diffuse		
	Fen, marsh & swamp	Point, diffuse, intermittent		
Air quality regulation	Arable & Horticultural	Intermittent	+2 (clean air but related to most other effects on services in this table)	Spreading measures will decrease NH_3 emission from fertilised land
	Improved grassland	Diffuse, intermittent		
Climate regulation	Arable & Horticultural	Diffuse, intermittent	-2 (risk of increased N_2O emissions}	Spreading measures will increase N_2O emissions from soil
	Improved grassland	Diffuse, intermittent		
	Broadleaved mixed and yew woodland	Point, intermittent	+2 (decreased N_2O emissions)	Lower N_2O emissions from forest soils
	Coniferous woodland	Intermittent		
	Calcareous grassland	Intermittent	+2 (decreased N_2O emissions)	Lower N_2O emissions from soils
	Fen, marsh & swamp	Point, intermittent		
Water regulation	Fen, marsh & swamp	Point, intermittent	+2 (reduced loss of water storage capacity)	Smaller loss of water holding capacity by mosses
Pollination	Calcareous grassland	Intermittent	+2 (loss of crop avoided)	Smaller impacts on flowering plants

Service	Habitat	Emission Type	Value (benefit/disbenefit)	Reason
	Fen, marsh & swamp	Point, intermittent		
Erosion regulation	Dwarf shrub heath	Diffuse	+2 (habitat preserved)	Reduction in heather canopy loss
	Fen, marsh & swamp	Point, intermittent	+2 (habitat preserved)	Smaller losses of peat forming mosses
Cultural services	Rivers & streams	Point, diffuse, intermittent	+2 (recreational and tourism, aesthetic, educational and cultural heritage benefits)	Slower rates of species loss and community change
	Standing open water & canals	Point, diffuse, intermittent		
	Broadleaved mixed and yew woodland	Point, diffuse, intermittent		
	Coniferous woodland	Diffuse, intermittent		
	Acid grassland	Diffuse		
	Calcareous grassland	Diffuse, intermittent		
	Neutral grassland	Diffuse		
	Dwarf shrub heath	Diffuse		
	Fen, marsh & swamp	Point, diffuse, intermittent		
Soil formation	Dwarf shrub heath	Diffuse	+2 (habitat preserved)	Reduction in heather canopy loss
	Fen, marsh & swamp	Point, diffuse, intermittent	+2 (reduced impacts on peat formation)	Smaller losses of peat forming mosses
Primary production	Broadleaved mixed and yew woodland	Point, intermittent	+2 (reduced toxic effects on vegetation)	Lower N toxicity from large NH_3 inputs
	Fen, marsh & swamp	Point, intermittent		
Nutrient cycling	Broadleaved mixed and yew woodland	Point, intermittent	-2 (reduce rates of nutrient cycling)	Reduction in large inputs driving N cycling
	Fen, marsh & swamp	Point, intermittent		

2.4 Towards a quantitative analysis

The NH_3 abatement measures of Scenario 1 are extended further by Misselbrook et al. (2007) to meet lower potential targets of 230 Kt NH_3 yr- (Scenario 2) and 210 Kt NH_3 yr^{-1} (Scenario 3) by introducing more stringent regulations for handling and storing manure (Table 2.5).

A full quantitative analysis for all ecosystem services in all habitats of these further measures to increase the level of reduction of NH_3 emissions from particular sources is impossible. This is because of the many gaps in knowledge and information that are discussed in more detail in the following stages of this report. We therefore propose a procedure based on the summary of effects in Technical Annex II, and in particular the most potential significant changes shown in Table 2.4, as the basis for identifying the most important areas of change under the more stringent measures. More rigorous quantitative analysis could then focus on these specific services and habitats in the first instance, as those that are likely to be most important.

Table 2.5 Scenario 2 and 3 NH_3 abatement measures (Misselbrook et al., 2007) and their emission types and pollution swapping potential (Misselbrook et al., 2008)

Abatement measure	Implementation steps	Emission reduction (Kt)	% reduction from baseline	Emission Type	Potential pollution swapping
More stringent storage requirements (scenarios 2 & 3)	• Cover manure with polythene sheeting during storage • Store all manure prior to spreading	14.8	6.6	Diffuse and point source	Greater potential for N loss during storage and spreading (NH_3, N run-off, nitrate leaching, N_2O emissions) Storage could result in further emissions of N_2O and methane
More stringent housing requirements (scenario 3 only)	• Use 25% additional bedding for cattle straw-bedded housing • Increase scraping frequency in dairy cattle cubicle houses • Increase the frequency of manure removal from layer hen housing with belt-removal systems (from once per week to twice-weekly) • Install air scrubbers to mechanically-ventilated pig housing • Frequent slurry removal from beneath slats (reduced in-house storage) for pig housing	24.2	10.8	Point source	Additional straw and air scrubbers have a low risk of pollution swapping Manure removal has a greater potential for N loss during storage and spreading (NH_3, N run-off, nitrate leaching, N_2O emissions)

In Table 2.5, abatement measures are classified by their source type. These can then be used in conjunction with those that had significant effects in Scenario 1 to identify which measures will have the greatest effect under the more stringent scenarios. For example, the implementation of Scenario 3 is largely concerned with limiting emissions from point sources (intensive animal units) and will therefore have an effect on those ecosystem services that were identified as most responsive to abatement measures on these sources. The services that are most likely on this basis to show a significant marginal change from Scenario 1 to Scenario 3 are shown in Table 2.6. The habitats affected are largely influenced by geographical distribution e.g. fen, marsh and swamp habitat is likely to show the greatest change under Scenario 3 due to its large overlap with pig and poultry as the dominant NH_3 sources in the south and east of England.

The next stage of analysis would then be to begin to subject effects on these services to more detailed quantitative analysis. Implementation of Scenario 2 results in a further emission reduction from diffuse sources of 3.5 to 6.6% reduction from baseline, roughly doubling the impact of abatement measures. By contrast Scenario 3 reduces housing emissions from a 1.4 to a 10.8 % reduction from baseline, a ten-fold decrease in point-source effects. Hence, moving from Scenario 1 to Scenario 3, although making a relatively small change in national emissions, would have a much greater proportionate effect on ecosystems services and in regions of the country for which point sources are most important.

Table 2.6.Ecosystem services most likely to be significantly affected in Scenario 3

Ecosystem service	Habitat	Strong marginal change from Scenario 1 to 3
Ecosystem goods	Rivers & streams	+
	Standing open water & canals	+
Water quality	Rivers & streams	+
	Standing open water & canals	+
	Broadleaved mixed & yew woodland	+
	Fen, marsh & swamp	+
Biochemical/genetic	Rivers & streams	+
	Standing open water & canals	+
	Broadleaved mixed & yew woodland	+
	Fen, marsh & swamp	+
Climate regulation	Broadleaved mixed & yew woodland	+
	Fen, marsh & swamp	+
Water regulation	Fen, marsh & swamp	+
Pollination	Fen, marsh & swamp	+
Erosion regulation	Fen, marsh & swamp	+
Cultural services	Rivers & streams	+
	Standing open water & canals	+
	Broadleaved mixed & yew woodland	+
	Fen, marsh & swamp	+
Soil formation	Fen, marsh & swamp	+
Primary production	Broadleaved mixed & yew woodland	+
	Fen, marsh & swamp	+
Nutrient cycling	Broadleaved mixed & yew woodland	-
	Fen, marsh & swamp	-

Any quantitative analysis would proceed, as in standard air quality management assessment, by applying the proposed emission reductions to point sources and using appropriate air chemistry and air transport models to predict the changes in the distribution of NH_3 concentrations and NHx deposition that were shown in Technical Annex II (Figures A3 and A4). These changes could then be overlain with the distribution of the relevant habitat area to assess the extent to which its exposure would be reduced. We note in passing here that in theory an ecosystem service based optimisation procedure could be used, in which the spatial distribution of key habitats could be used to maximise the benefit for ecosystem services of controlling particular point sources. However, this is at current state of knowledge a theoretical rather than a practical course of action.

The key questions are then whether we can quantify the effects of changes in NH_3 concentration and NHx deposition on specific ecological responses and hence ecosystem services, and then value the marginal changes in these services. These questions are considered in the following sections of the report.

2.5 Potential for the quantification of the impacts of policy options on specific ecosystem services

In order to quantify the impacts of changes in ammonia emissions on ecosystem services, it is first essential to establish a relationship between an environmental exposure and the ecological response. This relationship can then be applied in conventional air quality management by relating any planned changes in ammonia emission to levels of environmental exposure, either expressed as an air concentration or a deposition rate. Once this has been done, the exposure-response relationship can be used to estimate the change in ecological response, and hence ecosystem services, resulting from the policy change.

Here we consider the availability of relevant exposure-response relationships that might be used. This review is not intended to be exhaustive but rather to illustrate the issues that need to be considered. These relationships can come from a number of different sources:-

- Experimental studies, primarily in field chambers in which controlled concentrations of ammonia are applied;
- Experimental studies, primarily in the field, in which applications of wet deposited nitrogen are made, either in reduced or oxidised forms;
- Field transect studies around point sources, in which there is a clear source of ammonia, including experimental field release;

- Field surveys in which correlations are sought between current nitrogen deposition and spatial variation in either the current status of relevant ecological variables or changes in these over time (e.g. invasion of species adapted to higher levels of nitrogen);
- Computer models that simulate the long-term effects of ammonium on ecosystem structure and processes.

Field manipulation studies allow exposure-response relationships to be established under field conditions, and have been the most important source of information for setting critical loads of nitrogen. A detailed review of information from such studies is provided by Cunha et al. (2002). Although these provide an extensive source of data, establishing exposure-response relationships, they have a number of important limitations which need to be considered. The most important in the context of this exercise are that:-

- The background deposition to which experimental treatments are added is often poorly defined but is very important for interpretation. Several UK studies have found relatively little response to nitrogen addition, and it has been suggested this is because historical levels of nitrogen deposition have already had a significant impact on ecosystems services and biodiversity.

- The timescale of such studies is often rather short (typically 3 to 5 years), and hence may not be sufficient to detect long-term adverse effects of cumulative nitrogen inputs.

- Plot sizes are relatively small, and what happens on this scale cannot be automatically scaled up to an ecosystem scale

- The number of experiments is small, and site-specific factors mean that they may not be typical of the ecosystem or habitat of concern.

- Very few experimental studies have been conducted under field conditions using controlled levels of ammonia gas.

For these reasons, we have focussed our discussion of exposure-response relationships that can be used to quantify the benefits of ammonia emission reductions primarily on those using field survey data rather than experimental data. The data from field experiments provide mechanistic support for interpretation, and have been valuable in defining critical loads for nitrogen deposition (Bobbink et al., 2003) and critical levels of ammonia (Sutton et al. 2008), and so provide a supporting role. This has some analogy to the way in which assessment of the benefits of emission reductions for human health is largely based on epidemiological studies, supported by toxicological evidence where appropriate.

Relationships which could be used to quantify the benefits of ammonia emission reductions are discussed in detail in Technical Annex III, which covers climate and water quality regulation and changes in species numbers and composition. Annexe III is not intended to be an exhaustive review; rather we identify and discuss some relationships which have obvious potential, considering both their scientific strengths and weaknesses, and how they can be linked to ecosystem services. Quantifying climate regulation responses, and C storage in particular, to N deposition is problematic as relationships are often non-linear and depend on a wide range of factors. In contrast, relationships between N deposition and N leaching (water quality regulation) are generally more robust. However these are only well established for forests and cannot be applied to other ecosystems. In terms of changes in plant species composition, a number of relationships have been established between N deposition and species number, or the abundance of individual sensitive species. However these relate only to a limited number of habitats and species, and hence do not provide a basis for a comprehensive national scale evaluation. The results of the evaluation in Annexe III also highlight an important issue in quantifying the benefits of scenarios to reduce emissions from point sources of ammonia:- increased deposition, up to a critical threshold, could have significant benefits for ecosystem goods and climate regulation, through increased carbon uptake and sequestration, which are in principle quantifiable, but may have adverse effects on biodiversity which are much more difficult to quantify. Furthermore, the BAP and associated directives and legislation explicitly aim to devote significant resources to preventing successional change to protect biodiversity, but may also reduce productivity and carbon sequestration as a result.

Annexe III identifies a number of quantitative relationships that have been established between total nitrogen deposition and key responses such as ecosystem productivity and species richness. However, the specific requirement when considering the benefits of measures to reduce ammonia emissions is to quantify the benefits of reductions in concentrations or deposition of reduced, not total, nitrogen. Only for a very small number of specific ecosystem responses (e.g. the loss of particular sensitive species from wet heaths, or the loss of acidophytic epiphytic lichen species) can a robust relationship with NH_x inputs be established directly from experimental or field evidence. In other cases, the evidence suggests that, for the UK, the contribution of reduced nitrogen to these and other effects is likely to be greater than that of oxidised nitrogen. Hence, even using a conservative assumption that each unit reduction in NHx deposition causes the same response as indicated by the total nitrogen deposition, some valuation should be possible. If NHx nitrogen makes a bigger contribution to the observed dose-response relationship, then effectively the benefit for a unit reduction in ammonia emissions would be greater than that indicated by the relationship with total nitrogen deposition, and the value of the benefits would be underestimated using the relationships with total nitrogen deposition.

In conclusion, we would argue that, as long as the ecosystem services identified as being susceptible to NH_3 damage are susceptible to valuation through appropriate techniques, then extending a quantification of exposure-response relationships to valuation of the benefits of emission reductions should be possible. However, it is likely that this would only be a partial exercise, as we do not have comprehensive data for all ecological responses to ammonia emissions, and the associated ecosystem services, in all habitats.

2.6 Assess the effects on human welfare

The ecosystem services that are expected to be strongly affected by ammonia emission control measures under Scenario 1 (Misselbrook, 2007), across a range of broad habitat types, were determined qualitatively in Section 2.3. The broad habitat types and ecosystem services that may be most affected by the more stringent emission control measures (Scenarios 2 and 3) were also determined in Section 2.4. In this section we focus on the positive and negative outcomes for society that derive from these ecosystem services.

An important aspect of this analysis is making a clear distinction between the ecosystem services affected and the positive and negative outcomes for society that they provide. A recent Defra report (Haines-Young et al. 2007; NR0107) showed that different authors using different typologies do not always distinguish between ecosystem functions and services and societal benefits (e.g. De Groot, 1992; De Groot et al. 2002; MEA, 2005). Here, we explore the relationship between the services and the positive and negative outcomes for society by using the concept of an ‘ecosystem cascade’ which document how biophysical structures and processes give rise to ecological functions that in turn provide a service that potentially can be quantified in terms of positive and negative outcomes for society (Haines-Young et al. 2007). Table 2.7 shows these relationships for five of the strongly affected broad habitat types identified in Section 2.3.

The complexities and interrelationships between supporting, provisioning, regulating and cultural services, and issues of double counting, mean that seeking to place a value on each individual service may not be appropriate and ecosystems services may best be considered in combination rather than in isolation. In our study, the tabular approach replaces the mapping exercise suggested by Defra in its guide (Defra, 2007a) as a means of exploring the complex interactions between services and their end points. Double counting positive and negative outcomes for society is avoided by omitting the supporting services and the tabular format creates a framework whereby other double counting issues can be explored. The table also provides a starting point for identifying the trade-offs that would be involved in a full cost accounting (or full life cycle analysis) exercise e.g. N_2O emissions versus carbon sequestration in forestry, or higher N_2O emissions from spreading measures to arable land versus lower N_2O emissions linked to reduced deposition on other habitats. The table also highlights where effects may be additive e.g. reduced changes in species composition have benefits that can be economically valued in terms of ecosystem goods and cultural services.

Our qualitative analysis identified water quality and climate regulation as the services that are expected to be strongly affected across the broadest range of habitats. Table 2.7 shows the functional ecological relationships which would provide the basis for quantification of the effect, and the positive and negative outcomes for society that could be valued. Whether a value can be attached to these positive and negative outcomes will be considered in the next section.

Despite the fact that carbon sequestration in trees and soil was determined as a ‘positive effect’ rather than a ‘strongly positive effect’, in the qualitative analysis (Section 2.3), it is included in Table 2.7 as it is a potentially important effect. For example, our qualitative analysis shows that the implementation of Scenario 2 would have the effect of doubling the impact of abatement measures controlling diffuse sources, which could potentially reduce current carbon sequestration associated with forestry due to the fertilization effect of current NHx deposition. More generally, the increased impact of Scenario 2 measures would have the effect of enhancing the benefits identified across a range of habitats nationally.

Decreased N deposition, from point, diffuse and intermittent sources, direct to water bodies (i.e. rivers and streams, standing open water and canals) in general has a positive effect on water quality at national level and leads to societal benefits of clean drinking water and recreation, through preservation of habitats and species and fish numbers (angling). A similar effect is achieved via reduced N deposition to certain habitat types where leaching of nitrate to water bodies is reduced. However, these more widespread benefits have to be balanced at more local scales against the effect of spreading measures for arable and horticultural and improved grassland habitat types that decrease ammonia emissions but can increase N leaching to water bodies. An added complication is that the measures that affect nitrate leaching in these areas will also have implications for pollution swapping which could improve water quality at the expense of climate regulation (i.e. increased N_2O emissions). This complication also relates to landscape management decisions related to the Water Framework Directive (e.g. use of riparian buffer zones).

The beneficial effect of the relationship between NHx deposition and changes in species composition is also identified as a strongly positive effect across a broad range of habitats and the benefits manifest themselves in our analysis as maintenance of as-yet unused resources (biochemical/genetic characteristics of species) and cultural heritage aspects (recreational and tourism, aesthetic, educational and cultural benefits related to species diversity). This is an important area for consideration in this study as changes in species composition or changes in species numbers can have effects on ecosystem functions and services (see Balvanera et al. 2006) and could lead to double counting when these are considered alongside the direct effects of N deposition on ecosystem services.

The quantification of the benefits of reduction in ammonia emissions and deposition needs to be set in the wider context of how we manage our ecosystems in the UK. Many of the habitats that we value, and that are associated with species of high conservation value in the UK, are semi-natural habitats which are maintained by active management intervention to prevent succession, and maintain a diverse range of niches. Most of the exposure-response relationships with ammonia or nitrogen deposition discussed in Annexe III assume no active management, but in practice there may be very important interactions with management. For example, more active management by cutting, grazing or burning would be needed to maintain dominance of ericaceous shrubs in heathlands if N deposition was higher. Hence, in addition to the issue of the need for ecological restoration, and the associated management costs, ammonia emissions may lead to increased site management costs to maintain the current species composition. It is not clear how this can be incorporated within the Ecosystems Approach, as any costs of maintaining the desired services in terms of biodiversity would be hard to attribute specifically to ammonia emissions. It is possible that biomonitoring techniques that have been developed to identify sites with an impact of NHx deposition could be adapted for this purpose.

A considerable body of information already exists in the Netherlands, where a significant part of the country has been subjected to excessive concentrations of ammonia for several decades, but where reductions in emissions have been achieved, and active ecological restoration projects are underway, with governmental support. This provides a rich body of information which could be used to assess in more detail the specific impacts of ammonia on specific ecosystem services and their valuation (see Section 2.7).

Up to this point in the report, we have focussed specifically on ecosystem services in terms of the MEA (2005) definition introduced in Section 2.1. However, if the full implications of emission control measures for human welfare are to be considered in an Ecosystems Approach, as defined by Defra (2007b), we also need to consider other effects on human welfare, specifically through direct effects of changes in air quality on human health and on the built environment. These effects include:-

- direct effects of ammonia gas on human health;
- direct effects of ammonium or nitrate aerosol on human health;
- effects of odour;
- effects of gaseous ammonia or ammonium aerosol on the built environment.

While gaseous ammonia can have adverse health effects at high concentrations, these are orders of magnitude higher than expected environmental concentrations, and hence are only of concern in an occupational setting. In contrast, there is extensive evidence of the effect of fine particles, to which secondary ammonium nitrate and ammonium sulphate aerosol contribute, on respiratory and cardiac disease. There is substantial uncertainty in both the modelling of the relationships between primary gaseous ammonia emissions and nitrate and ammonium concentrations, and the contribution of different chemical components of final particles to health outcomes, which are outside the scope of this report. However, an assessment of effects of a 30% reduction in UK NH_3 and NH_4 emissions on secondary aerosol and hence on $PM_{2.5}$ concentrations has recently been conducted for Defra (Stedman, 2008).

Table 2.7 Examples of potentially significant positive and negative outcomes linked to ecosystem services of broad habitat types under Scenario 1

Note: *Negative effects in italics*

Broad Ecosystem type	Effect of abatement measures	Source	Biophysical structures affected	Direction of effect (+ve / *-ve*)	Potential for pollution swapping	Intermediate services affected	Positive and negative societal outcomes
Arable and horticultural	Spreading measures leading to reduced NH_3 emission and deposition, leaching or N_2O emission	*Diffuse/Intermittent*	*Water*	*-ve depending on land management*	*yes*	*Relationship between spreading and nitrate leaching*	*Fishing, biochemical /genetic and drinking water Recreation and tourism*
		Diffuse/ Intermittent	*Atmosphere*	*-ve*	*yes*	*Relationship between spreading measures and N_2O emissions from soil or from riparian buffer zones*	*Increased contribution to climate change*
		Intermittent	Atmosphere	+ve	yes	Relationship between NHx emissions and deposition	Effects on other ecosystems via deposition
Rivers and streams	Reduced NH_3 and NH_4 deposition direct to water bodies	Point, diffuse, Intermittent	Water	+ve	no	Relationship between NHx deposition and N concentration and pH of water; Relationship between nitrate concentration and pH of water and changes in species composition	Fishing, biochemical /genetic and drinking water (provisioning service) Recreation and tourism
Broadleaved mixed and yew woodland	Reduced NH_3 / NH_4 deposition	*Point, diffuse, Intermittent*	*Soil and vegetation*	*-ve*	*no*	*Relationship between NHx and tree growth*	*Loss of production and biomass*
		Point, Intermittent	Soil and vegetation	+ve	no	Relationship between NHx deposition and changes in species composition	Biochemical/genetic resources preserved
		Point, Intermittent	*Atmosphere*	*+ve/ -ve*	*no*	*Relationships between NHx deposition and N_2O emissions from soil, and carbon sequestration in trees and soil*	*Altered contribution to climate change*
		Point, diffuse, Intermittent	Soil and vegetation	+ve	no	Relationship between NHx deposition and changes in species composition	Recreational and tourism, aesthetic, educational and cultural heritage benefits

Broad Ecosystem type	Effect of abatement measures	Source	Biophysical structures affected	Direction of effect (+ve / *-ve*)	Potential for pollution swapping	Intermediate services affected	Positive and negative societal outcomes
Calcareous Grassland	Reduced NH_3 / NH_4 deposition	Intermittent	Soil and vegetation	+ve	no	Relationship between NHx deposition and changes in species composition	Biochemical/genetic resources preserved
		Intermittent	Atmosphere	+ve	no	Relationship between NHx deposition and N_2O emissions from soil	Reduced contribution to climate change
		Intermittent	Crop	+ve	no	Relationship between NHx deposition and flowering Relationship between flowering and pollinator abundance	Loss of crop avoided
		Diffuse and Intermittent	Soil and vegetation	+ve	no	Relationship between NHx deposition and changes in species composition	Recreation and tourism, aesthetic, educational and cultural heritage benefits
Fen,marsh and swamp	Reduced NH_3 / NH_4 deposition	Point, diffuse, intermittent	Soil and vegetation	+ve	no	Relationship between NHx deposition and changes in species composition	Biochemical/genetic resources preserved
		Point, Intermittent	Atmosphere	+ve	no	Relationship between NHx deposition and N_2O emissions from soil	Reduced contribution to climate change
		Point, Intermittent	Soil and vegetation	+ve	no	Relationship between water holding capacity of mosses and NHx deposition	Increased water retention
		Point, Intermittent	Soil and vegetation	+ve	no	Relationship between NHx deposition and cover of peat forming mosses	Reduction in erosion
		Point, diffuse, Intermittent	Soil and vegetation	+ve	no	Relationship between NHx deposition and changes in species composition	Recreation and tourism, aesthetic, educational and cultural heritage benefits

The technical basis of this assessment of benefits for human health can be summarised as follows. Source-receptor models are first used to simulate the change in secondary inorganic aerosol (SIA) concentrations in each grid square across the country in response to the change in NH_3/NH_4 emissions. These relationships are quite uncertain, and are strongly influenced by climatic factors. They are also non-linear, especially in winter. It is assumed that SIA concentrations contribute to 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 is assumed to be linear. On this basis the 2008 study for Defra does not deem it necessary to simulate detailed spatial and temporal variation in human exposure; instead, the resident population in each of the grid square is used to calculate a change in the population-weighted annual mean concentration, and hence using a simple exposure-response relationship, together with estimates of disease prevalence, to calculate the health impacts across the country. Recent estimates (Stedman, 2008) suggest that the population-weighted annual mean $PM_{2.5}$ concentration would reduce by less than 0.2 µg m^{-3} as a result of reducing NH_3 emissions by 30% from their 2006 levels (from 311 kTonnes to 217.7 kTonnes).

Important societal benefits of reducing ammonia emissions, linked to reduced formation of ammonium containing particulate matter may also arise from improved atmospheric visibility, in addition to effects on health. At a global scale they also contribute to the cooling effect of ammonium sulphate aerosol in the atmosphere that counteracts global warming (Ramanathan and Feng, 2008). However, we are not aware of any attempt to value these effects, although it may be theoretically possible e.g. effects of hazy days on tourism and relating the cooling effect of aerosols to the shadow price of carbon. Regarding the built environment the authors are unaware of any research that shows significant effects of ammonia on man-made materials and cultural heritage at concentrations found in the UK. Evidence does exist for effects at higher concentrations e.g. controlled chamber studies have shown that at high concentrations (100 -200 ppm) aerial ammonia reduces the corrosion rates for steel (Zhu et al. 1999).

2.7 Valuation of changes in ecosystem services

2.7.1 Challenges and opportunities for valuation

A recent review for Defra of ammonia damage costs concluded that no study had identified an ammonia damage cost related to ecosystem impacts (Entec, 2007). The Entec report identified a number of general attempts to value the externalities of agricultural activities, primarily through contingent valuation methods. The earlier sections of our report identify ecosystem services susceptible to NH_3 damage and suggest that, in some cases, dose-response relationships exist which could make valuation of the benefits of reductions in ammonia emissions possible. However, it is likely that this would only be feasible currently for a subset of the affected ecosystem services in a subset of the affected habitats.

The ecosystem service / habitat type combinations assessed for valuation possibilities follow directly from the categorisation and summary presented in Table 2.7, arising from the changes predicted to result from ammonia emission control under Scenario 1. In the valuation discussions which follow we focus on changes to ecosystem service delivery in the following habitat types of Table 2.7: arable and horticultural; rivers and streams; broadleaved, mixed and yew woodland; calcareous grassland; and fen, marsh and swamp. These habitat types are predicted to display a sufficient range of ecosystem service effects to illustrate the challenges and opportunities which adopting an ecosystem service approach affords for environmental valuation.

It is acknowledged that benefits transfer will have to be used extensively in ecosystem service-based valuation (Defra, 2007a). Benefits transfer is "a process by which the economic values generated [from changes in ecosystem service delivery] in one context – the 'study site' – are applied to another context – the 'policy site' – for which values are required" (Defra 2007a, point 4.30, p38).

Considerable care needs to be exercised, however, in determining whether a particular valuation study can provide an appropriate basis for benefits transfer-based valuation of the changes in ecosystem service delivery arising from a proposed change in ammonia management policy. This is because it is usually the value associated with a ***marginal change*** in ecosystem service delivery which is sought. To illustrate this, consider the example of the value arising from the marginal recovery of species diversity in calcareous grassland as a consequence of implementing ammonia control measures. The key question is whether the study which will form the basis for benefits transfer contains a valuation function which explicitly recognises the marginal contribution which the ecosystem outcome concerned (species diversity in this case) makes to the delivery of the ecosystem service which we are aiming to value (in this case, recreational, heritage and aesthetic components of the cultural service). If a functional linkage of this type is not present in the original study then it is not a satisfactory starting point from which to begin a benefits transfer exercise. It is therefore not sufficient simply to know the value delivered by a day's recreation on a nature reserve in limestone grassland; to implement an ecosystems services-based valuation of ammonia control policies we would need to know how the value of that day's recreation would ***change*** when the species diversity of the nature reserve increases as a result of implementing ammonia control measures. The required valuation could only be produced if species diversity (or an appropriate proxy) featured explicitly in the valuation function of the starting point study, and, ideally, if the valuation function in the starting point study had been demonstrated to be 'robust', perhaps by showing that a valuation function developed on one half of the dataset from the study provided a good estimate of the values actually reported in the other half of the study dataset (Brouwer 2000, Brouwer & Bateman 2005).

Annexe IV evaluates and discusses possible valuation methods for changes in ecosystem service delivery in the habitats listed in Table 2.7. This detailed evaluation shows that the type of difficulty discussed above is encountered frequently when considering specific ecosystem services. The analysis in Annexe IV also indicates that valuing changes in the delivery of ecosystem services arising from changes in ammonia management policies will present a considerable challenge at present. This challenge arises for a number of reasons. In many cases the relevant dose-response relationships and economic parameters (such as price elasticity of demand and industry-wide cost functions, for example) are not readily available for valuing changes in provisioning or regulating services. With regard to changes in the delivery of cultural services, the marginal impact of ammonia-driven changes in environmental quality has not typically been included as an explicit term within the valuation functions of earlier studies, making appropriate benefits transfer from the literature very difficult indeed.

However, these challenges are not specific to the use of the ecosystem service approach for valuing changes in ammonia management; they would apply similarly to attempts to apply the ecosystem services approach to value changes in service delivery arising from proposed policy changes in almost any other area of policy interest too - flood risk management, agricultural practice and managed retreat to name but a few. Indeed, changes in ecosystem service delivery from ammonia management are rather better served by existing valuation studies of cultural service delivery than many other policy areas because of the effort that has been invested recently into nitrogen-related changes in water quality under the requirements of the Water Framework Directive.

The challenge presented by application of the ecosystem services approach is considerable, but so is the potential opportunity which the approach affords for capturing the wide spectrum of values delivered by ecosystem processes and functions. These opportunities will only be realised, however, if sufficient effort, commitment and ingenuity are applied to address the initial challenges of adopting an ecosystems service based approach to valuation.

2.7.2 Comparison of human health and ecosystem service benefits

Annexe IV only considers ecosystem services. While it identifies a number of major challenges to valuation of the benefits of ammonia emission control for these services, it also identifies considerable potential for quantification of these benefits. It would hence be useful to ask the question: how does the size of the benefits of ammonia emissions control for these ecosystem services compare with those estimated for human health?

The health benefits of ammonia emissions reductions have been calculated by Watkiss (2008). The damage costs are applied to the marginal emission changes over time before discounting in a standard appraisal framework. The main health effect of the decreased $PM_{2.5}$ concentrations, in economic terms, was a decrease in mortality expressed as a decrease in the years of life lost (YOLL). Using a standard valuation of the YOLL, and a time lag of 40 years between current implementation of the proposed changes in ammonia management and the consequent reduction in mortality, a benefit value of £1407 per tonne of NH_3 emission reduction was calculated. Since, a 30% reduction corresponds to a reduction of 94 kTonnes, based on 2005 data, this would give an estimated total benefit of £131 million to human health from measures which delivered a 30% reduction in ammonia emissions.

Would the benefits of this scenario change significantly if account was taken of impacts on ecosystem services? A rough comparison of the scale of the financial impact of ammonia reduction on human health compared with the scale of financial impact on ecosystem services can be obtained by considering changes in the climate regulation service costed with Defra's standard shadow price of carbon (SPC) methodology (Defra 2008[1]). Section 2.5 and Annexe III indicate that a reduction in ammonia emissions is likely to affect the climate regulation service through two main mechanisms (i) by reducing tree growth in forest and woodland ecosystems and thus reducing the quantity of carbon sequestered, effectively increasing equivalent CO_2 emissions (CO_2e), and (ii) by changing the relationship between NH_x deposition and N_2O emissions from soil or riparian buffer zones in a range of different habitat types. N_2O emissions from arable and horticultural habitats are considered likely to increase as a consequence, whereas

[1] Refer to Defra note on costings using the shadow price of carbon available on the web at: www.defra.gov.uk/environment/climatechange/research/carboncost/

emissions from calcareous grassland and fen. marsh and swamp habitats are considered likely to decrease. We will produce rough estimates of the ranges of costs and benefits arising from these two mechanisms in turn.

(i) Annexe III explains that the best estimates of the reduction in carbon sequestration in forests and woodlands vary between 30-70 kg C kg^{-1} deposited N. Taking a best estimate of 50 kg C sequestered kg^{-1} deposited N and knowing that 1 tonne of carbon sequestered is equivalent to 3.67 tonnes CO_2e, this is equivalent to 183.5 tonnes of CO2e for every tonne of deposited N. Using Defra's 2008 value for the SPC (£26.5 $tonne^{-1}$ CO_2e) this reduction in carbon sequestration represents a cost of £4863 ha^{-1} $year^{-1}$ for forest and woodland ecosystems per tonne reduction in N deposition. This estimate does not consider effects on carbon sequestration in soils. To use this estimate to determine the benefit of ammonia emission reductions, other information would be needed which is outside the scope of this project. This would include the modelled relationship between changes in ammonia emissions and N deposition to UK woodlands (analogous to the relationship between ammonia emissions and concentrations of secondary inorganic aerosol for health impacts). However, assuming that all the 30% reduction of 94 ktonnes of emissions were converted to reduced N deposition in the UK (which is not unreasonable given the relatively small transport distance of ammonia) and that forests and woodlands cover 10% of the UK land area, the cost of the reduction in ammonia emissions would be £46 million in total annually.

(ii) Annexe III also indicates that N_2O release from soils and riparian buffer zones is typically 1% of N deposition nationally, but could be between 6% and 25% of ammonia deposition in some locations. In addition, some specific slurry management techniques to reduce ammonia emissions could have much larger local benefits for local N_2O emissions. For the purpose of an illustrative calculation, let us assume a value of 5%, i.e. 50kg of N_2O emissions per tonne of ammonia deposited. N_2O has a global warming potential 310 times that of CO_2, hence the CO_2e of these changes in N_2O emissions is 15.5 tonnes per tonne of change in ammonia deposition. Costing these changes using Defra's 2008 value for the SPC (£26.5 / tonne CO_2e) produces a value of £411 per tonne of change in ammonia deposition. Assuming, as above, that a 30% reduction in emissions gives a 30% reduction in deposition across the country, then the total annual value would be £39 million, assuming the 5% value applied nationally. In practice, the UK-wide picture is much more difficult to calculate because N_2O emissions are likely to increase from some habitat types and decrease from others, and depend on the emission control measures that are applied. As for woodland carbon sequestration, spatial summation of the areas of each affected habitat type would be required to take these calculations further.

Overall, however, it is clear that the scale of benefits and costs arising from changes in the provision of the climate regulation service are likely to be of at least of the same order as those arising from the human health impacts of changes in ammonia emissions. Indeed, if as some experts believe, secondary aerosol makes a less significant contribution to health effects than other types of particles, these climate regulation values could be relatively more important. Furthermore, our assessment relates only to the climate regulation services, and does not consider the full range of ecosystem services identified in Table 2.7 and discussed in Annexe IV.

It is very important to emphasise here that we have only made a very preliminary estimation based on an assumed linear relationship between emissions and deposition, with no consideration of the timescale of ecological responses. Since responses of relevant chemical and biological variables may lag significantly behind reductions in deposition, this is an important omission in the calculation. Furthermore, evaluation of health benefits can be based on changes in national mean population-weighted concentrations, because it is assumed that all people of a given age respond to changes in aerosol concentrations in the same way. As we have already emphasised, the spatial distribution of habitats of different sensitivity relative to different types of emissions sources, and the fact that emission reductions will have a greater effect on deposition close to source, means that a much more detailed spatial analysis would be needed to value ecosystem service benefits compared to health benefits. The implications of this for air quality management are considered further in the next section.

3. Comparison of the Ecosystems Approach with existing air quality management (Objective 2)

3.1 Introduction

The analysis described above for Objective 1 (Section 2) provides an outline of an ecosystem services framework which could be applied for assessment of options for control of ammonia emissions. However, to assess the implications for policy delivery of this approach, it is important to understand how it differs from the air quality management approaches for ammonia that are currently applied, and what potential there is for the Ecosystems Approach to add value to existing approaches.

An overview of UK air quality management approaches, and their specific application to NH_3 and to impacts on ecosystems is provided by the UK's Air Quality Strategy (Defra, 2007c). In generic terms, the procedure adopted is the same as that for the air quality aspects of an Ecosystems Approach (Defra, 2007a). A baseline is first constructed to provide the best estimate of the current and future concentrations of the pollutants of concern, assuming that all policies and commitments that are in place are implemented as planned. The consequences of a range of additional policy options for future concentrations are then modelled, and the benefits of each option are then assessed. The key question, however, is how effects on ecosystems, and those of ammonia in particular, are considered within this framework.

The National Air Quality Strategy (NAQS) recognises that ammonia can contribute to effects on human health, on ecosystem eutrophication and acidification, and that it has direct toxic effects on plants. Air quality objectives for eutrophication and acidification are defined in terms of critical loads, while direct toxic effects are defined in terms of a critical level. The NAQS recognises the importance of protection of sites and ecosystems of high conservation value, driven partly by other legislative drivers such as the Habitats Directive, when assessing the significance of exceedance of these air quality objectives. Values of critical loads and critical levels are set within the CLRTAP, based on evaluation of current scientific knowledge, and are then applied in the UK. In addition, Integrated Pollution Prevention and Control (IPPC) provides an important regulatory framework for protecting designated sites which is particularly significant for NH_3 hot-spots from large point sources such as pig and poultry farms.

However, the detailed technical and quantitative assessment of the costs and benefits of different policy measures which forms the basis of the NAQS has focussed on human health, with only a small number of environment benefits (e.g. effects of ozone on crop yield) considered quantitatively. Hence, benefits assessment in terms of ecosystem services has not to date been integrated into the NAQS. Defra (2007c) 'support the development of cost effective policies aimed at reducing critical load exceedances' but to date tools to evaluate the benefits of reducing critical load exceedance have not been applied. Defra (2007c) also recognise that, for ammonia, policy on emissions to air needs to be placed in a wider policy framework and suggest that 'a holistic and strategic approach will be taken to tackling nitrogen emissions through nutrient management in the wider context of losses to water, air and impacts on climate change', but do not identify any specific tool to achieve this goal. Finally, integrated assessment of benefits for human health, the built environment, and ecological services is needed to implement a full cost-benefit analysis. In this section, we consider how an Ecosystems Approach, and specifically a valuation of ecosystem services, could contribute to these policy aims, drawing on the detailed analysis in Section 2.

Defra (2007c) identifies no exceedance of the critical level of 8 µg m^{-3} as an annual mean concentration in the UK. However, this is based on a 1km^2 mean concentration, and hence ignores the possibility of more local effects. Furthermore, a recent proposal has been adopted within CLRTAP which would lead to a significant reduction of the critical level to 1 or 3 µg m^{-3}. Critical loads for effects on eutrophication and acidification are, in contrast, exceeded in many areas of England and Wales. In assessing the implications of any future policies to further control emissions of ammonia, as well oxides of nitrogen and sulphur, reduction in the extent of critical load exceedance (both in terms of land area and the degree of exceedance) is an important driver for the NAQS. But, what benefits would reducing exceedance of critical loads (or critical levels) have for ecosystem services?

3.2 Environmental thresholds and ecosystem services

Both critical loads and critical levels are set to prevent 'significant harmful effects on specified elements of the environment'. In other words, they provide policy makers with values of ecological thresholds above which adverse and potentially irreversible environment effects may occur. Critical loads are typically calculated using a steady state mass balance approach to determine, at steady state, the rate of deposition at which a critical chemical threshold for effects is exceeded. Alternatively, for eutrophication effects, an empirical critical load has been set based on observed effects in the field and in long-term field experiments.

A key question for the application of the Ecosystems Approach is then how the criteria used to set the values of critical loads and critical levels relate to the ecosystem services provided by the habitats in question. We identify three different answers to this question.

1. Steady state mass balance critical loads

Steady state mass balance critical loads are not linked to specific ecosystem processes or services. Rather they calculate the nutrient balance by estimating inputs and outputs, aiming to prevent accumulation of nitrogen and exceedance of a critical chemical threshold, expressed as a soil solution concentration. Hence the key question is whether this chemical threshold has any clear relationship to a specific ecosystem service. The chemical thresholds are set largely to prevent specific changes in species composition, for instance from blueberry to grass, and from heath to grass. The challenges of relating such specific changes in species composition to ecosystem services were discussed in detail in Section 2.5 and Annexe III, but a clear general conclusion is that exceeedance of these critical loads cannot readily be related to ecosystem services.

2. Empirical critical loads

For empirical critical loads to prevent eutrophication, a range of adverse effects were identified as potentially occurring when the critical load is exceeded. Table 3.1, taken from Bobbink et al. (2003), lists these for the specific habitats. While there is considerable variation between habitats, these effects can generally be characterised as one of three major classes of impacts:-

- Invasion of competitive, fast growing species
- Decreased plant species diversity or loss of characteristic species of the habitat,
- Increased nitrate leaching once the system reaches nitrogen saturation.

This is broadly consistent with the major baseline impacts of ammonia emissions which we have identified using the ecosystem services framework. However, the critical load approach does not consider the implications of loss of characteristic species or nitrate leaching in terms of specific ecosystem services – rather it is simply set to prevent these adverse effects. Furthermore, changes in primary production are treated quite differently under the Ecosystems Approach than under the critical load approach. Whereas the former sees this an increase in provisioning services, the latter sees this as an adverse effect because it is normally associated with increased cover of fast-growing species which will out-compete other valued species for the particular habitat. The balance between these two effects is habitat specific – for most woodlands and grasslands, for example, primary production is a central ecosystem service, but for mires or sand dunes it is not. A further difference lies in the issue of climate regulation, which is identified as a critical ecosystem service in our analysis, but is not considered explicitly in setting critical loads, as these do not consider effects on carbon sequestration. The critical load approach also only considers specific sensitive habitats, and does not provide a comprehensive assessment of the impacts of NHx deposition on all major habitats. Hence, while there are some common elements between the effects of critical loads exceedance and the ecosystem service analysis in this report, it is clear that an assessment based only on critical load exceedance does not capture all the effects of ammonia emission control on ecosystem services.

3. Critical levels

For critical levels of ammonia, values have been set to protect the most sensitive organisms for which there is concrete evidence of adverse effects. Critical levels for use within the CLRTAP have recently been revised after a workshop in December 2006. Two values were set, as annual mean concentrations:-

- 1 μg m^{-3}, to prevent adverse effects on lichens and bryophytes were these are 'important for ecosystem integrity'

- 3 μg m^{-3} to prevent adverse effects on woodland ground flora.

These values were set based on transect studies around point sources, supported where possible by controlled experiments. It is important to note that these changes were not defined to protect individual species – rather the key response is a shift from, for example, a community of acidophytic species to one dominated by nitrophytic species (Wolseley et al. 2006). The question of how such effects relate to ecosystem services in different habitats was not specifically considered, but rather the aim was prevent loss of sensitive groups of species. Hence, critical levels only consider loss of characteristic species and not the wider impacts of ammonia emissions.

Table 3.1. Effects of nitrogen deposition for which there is a reliable or quite reliable basis for setting a value for the critical load (adapted from Bobbink et al., 2003). Note that only broad habitats that are sensitive to increased nitrogen deposition are included.

Ecosystem	Effects of critical load exceedance
Broadleaved, Mixed and Yew Woodland	Increased nitrogen mineralisation in soils Increased risk of nutrient deficiencies in trees Changes in ground flora species composition, including increase of nitropholous species Reduced mycorrhizal species composition
Coniferous Woodland	Increased nitrate leaching Increased risk of nutrient deficiencies in trees Changes in ground flora species composition, including increase of nitropholous species Reduced mycorrhizal species composition
Acid Grassland	Increase in graminoids Decline in typical species
Calcareous Grassland	Increased in tall grasses Decline in diversity Increased N mineralization and N leaching
Neutral Grassland	Increase in graminoids Decline in typical species
Dwarf Shrub Heath	Transition of heather to grass Decrease in lichens
Bogs	N saturation and reduction of peat mosses Increase in sedges and vascular plants
Montane Habitats	Effects on bryophytes and lichens
Supralittoral Sediment	Increase in tall grasses Increase in N leaching Decrease in prostrate plants

In summary, critical loads and levels are set to prevent a defined set of adverse effects, most of which are specific for the habitat or species change of concern. National policy has focussed on reducing the exceedance of critical loads, largely because these are still exceeded over large areas. In contrast, there has been relatively little concern about the critical level because it applies to point sources. Where national policy has considered ecosystem effects, it has considered the area of critical load exceedance, or an index of a combination of the area and degree of exceedance. The implications of this exceedance for ecosystem services are not considered. Use of the Ecosystems Approach, based on the ideas outlined in this report, would clearly allow more explicit consideration of the consequences of critical load exceedance, and hence would add a new dimension to policy evaluation.

However, it might also be questioned whether an approach based on environmental thresholds can actually be integrated into an Ecosystems Approach. Many ecologists have argued that the thresholds used are artificial, and are based on limitations of experimental design or statistical analysis, or the limited timescale of the studies. They argue that since nitrogen is an essential nutrient, there is in reality no threshold for an ecological response and that the proposed critical loads and limits are artefacts of our limited scientific knowledge. Hence, it could be argued that valuing the benefits for ecosystem services of ammonia emission reduction should be based on exposure-response relationships with no threshold. Such an approach would be analogous to the approach used to value the health benefits of ammonia emission control, through the contribution to secondary aerosol formation, and hence would provide a consistent methodological framework for an integrated analysis by application of the Ecosystems Approach.

A further challenge to ecological and economic valuation of the benefits of ammonia emissions control is that of timescale, since soil and water concentrations will not respond immediately to changes in emission and deposition, and plant and animal responses may lag further. For valuation of health impacts, the same issue of timescale exists, albeit in a different form, especially when quantifying and valuing long-term effects of particulates on longevity. Within the NAQS, an empirical approach based on a timescale of 40 years has been adopted. Dynamic models which predict the dynamics of soil and water concentrations, and hence species or community presence, in response to changes in nitrogen deposition are now reaching the stage where application for policy application is feasible (de Vries et al., 2008). Such models will explicitly predict how the processes underlying ecosystem services, such as carbon sequestration, nitrate leaching and species composition will change over time. Hence the potential is emerging of an integrated ecosystem approach to valuing the benefits of reduced ammonia emissions, based on exposure-response relationships and dynamic models, which could replace the current approach based on environmental thresholds.

3.3 Valuing ecosystem service benefits and policy integration

One major reason that ecosystem service effects have not been included in NAQS to date is the perceived lack of information for valuing the benefits (e.g. Entec, 2007). However, the analysis which we presented in Section 2 suggests that this is a pessimistic conclusion; exposure-response relationships and valuation methods exist that would allow such an evaluation to begin to be developed, through the use of the Ecosystems Approach. Clearly, the sheer range of different effects in different habitats and on different species creates an enormous barrier to valuation. However, the difficulty in valuing ecosystem effects may not be the level of assumption that is needed (which may be no greater than those used to value benefits for human health), but the lack of standardised and agreed methods to do so. Assessment of the benefits for human health relies on putting values on a human life, or on ill health, through standard measures such as the VOLY (Value of Life Year). The potential value of the Ecosystems Approach is that it allows a wide range of ecological effects to be considered in terms of a narrower suite of ecosystem services – i.e. it creates a limited range of outcomes which can in principle be valued.

A further aspect which is strongly emphasised in the NAQS (Defra, 2007c) is the need to integrate ammonia emission control into the wider context of nutrient management. Polices implemented under the Water Framework Directive and the Nitrates Directive will affect the management of nitrogen and of fertilisers and animal wastes and hence influence ammonia emissions, while the reform of the Common Agricultural Policy may also influence emissions. Of particular relevance is the implementation of Catchment Sensitive Farming schemes which provide a framework for integrated analysis of impacts on ecosystem services at a catchment scale, including those arising from emissions to air. The challenge will be to use the Ecosystems Approach effectively to link such a catchment-specific approach with the aim of meeting national emissions targets. This implies addressing the challenge of identifying the spatial distribution of measures to reduce ammonia emissions that would maximise the benefits for ecosystem services.

3.4 The spatial dimension

The NAQS provides a clear framework for national evaluation of air quality management, based on assessing the costs and benefits of different policy interventions. It has an explicit spatial element, depending on whether there is a threshold for adverse effects on human health. Where there is such a threshold (e.g. for NO_2), the prime aim is to reduce 'hotspots' of high concentrations; where there is not (e.g. for PM_{10}) the prime aim is reduce overall population exposure. For ecosystem effects, there is currently a threshold driver, with the aim of reducing exceedance of critical loads for different habitats. However, if a spatial strategy for ammonia was based on ecosystem services rather than critical load exceedance, the priorities in terms of both the location and techniques used for emission control could be quite different because :-

1. There are potential positive and negative outcomes for society of ammonia emissions on ecosystem services below a particular threshold;
2. The point at which adverse effects on ecosystem services dominate any positive effects, for example, on production or carbon sequestration, is habitat-specific, and indeed even site specific.

Much more detailed analysis would be required to identify those sources and locations where control of ammonia emissions would have the greatest benefit for national or regional ecosystem services rather than for critical load exceedance, and much of the information needed for such an ambitious national implementation of the Ecosystems Approach is unavailable. However, we would argue for three reasons that this should not prevent further consideration of the spatial dimensions of the Ecosystems Approach in air quality management, since it may provide opportunities to optimise the benefits of measures to reduce overall emissions to meet a national emission ceiling.

Firstly, our work has clearly identified some measures (e.g. various controls on emissions from large pig and poultry units) which would have local benefits for ecosystem services, as well as those arising from the regional reductions in emissions from point sources which were the focus of the analysis in Section 2. These local benefits are unlikely to arise from other measures (e.g. relating to fertiliser applications), which have no single point source. Hence, any national strategy must consider the impacts of point sources and focus mitigation on these sources if it is to maximise the benefits for ecosystem services (although it will also need to consider the ‘pollution swapping’ implications). Such an approach could be linked to application of critical levels to identify locations where direct impacts of ammonia from point sources are most significant. Currently, it is argued in the NAQS that ammonia critical levels exceedance need not be considered because these were not exceeded as 1km^2 averages. However, critical levels for ammonia have now been reduced significantly. The implications for exceedance have yet to be fully evaluated in the UK, but they are likely to show exceedances in many 1km squares.

Secondly, as we demonstrated in Section 2.3, the spatial approach can operate at a broader regional scale, by combining information on the relative importance of different sources and the relative importance of different habitats. Hence the analysis presented in this report already demonstrates how a broad qualitative analysis can be used to assess where emission control would have the greatest benefit for ecosystem services.

Finally, the Environmental Agency (EA) has been undertaking a pilot study of the possibility of basing regulation of point sources on ‘Air Quality Outcomes’, which relate to the biosphere around the point source in question, rather than using emission limits or air quality objectives set as concentrations. Bealey et al. (2008) clearly differentiate between a biological effect or impact, and an ‘outcome’. While the former relate to any observed biological effect, an ‘outcome; is defined as a significant (‘costly’ in the broadest sense of the term) change in the function or structure or an organism or community. Examples of such effects are a degree of irreversibility and a significant cost related to the outcome. This distinction clearly mirrors that between an ecological effect and the implications for ecosystem services which was discussed in Section 2. Application of ‘outcome’ based regulation would imply different levels of emission control on individual point sources depending on the sensitivity and significance of the surrounding habitat. It also could readily be integrated with the application of the Ecosystems Approach to management of ammonia emissions.

In summary, the strengths that this study have identified for the application of the Ecosystems Approach to air quality management can be related to Defra’s key principles in the following way:

- It provides a framework for systematic assessment of ecological effects; it allows information on specific species, process and habitats to be synthesised;
- It captures important regulating effects and can highlight pollution swapping potential, such as those for water quality and climate regulation, which are not currently considered in the NAQS;
- It can be adapted to include consideration of effects at various scales and at a national level;
- It can support the definition and application of environmental limits by defining the specific ecosystem services which they can protect;
- It provides a framework for a comprehensive holistic assessment, involving human health, ecosystem services, and water quality, and nutrient budget management;
- It provides a framework for adaptive management, since the implications for different ecosystem services of alternative emission control policies, and the possible tradeoffs involved, can be clearly identified.

The weaknesses of the Ecosystems Approach mainly relate more to its generic application, rather than to the specific context of air quality management, for which specific benefits are identified by our analysis. These generic issues will be considered as barriers to implementation in the next section.

4. Barriers to implementation of Ecosystems Approach for Ammonia in the UK (Objective 3)

4.1 General barriers of implementing the Ecosystems Approach

The Defra commissioned study (NR0107) on England's Terrestrial Ecosystem Services and Rationale for an Ecosystem-Based Approach (Haines-Young & Potchin, 2007) states that the principles underpinning the Ecosystems Approach are consistent with the current UK Strategy for Sustainable Development but that it is not clear how Defra's vision could be used operationally. In this study three general barriers to implementation have emerged, which although not insurmountable, must be addressed if the ecosystem services approach is to be successfully applied in the UK.

(i) Treatment of the effect of air quality on human health

We have, as explained in Section 2.1, separated analysis of the effects of ammonia emissions from agricultural practices on atmospheric aerosol concentrations, and hence on human health, from the effects of ammonia/ammonium deposition on ecosystem services which are the focus of this report. The effects of ammonium aerosol on human health are considered in national air quality management, alongside effects on climate and visibility.

However, one commentator on our report argued that:-:

'The report considers that the effect of ammonia, for example, on carbon sequestration can be calculated as a change in delivery of the ecosystem service of carbon sequestration i.e. the ecosystem provides some level of sequestration in the absence of ammonia pollution, and a different amount of sequestration when ammonia is present. We can then calculate the value of this change in ecosystem service. By analogy, the ecosystem provides a certain level of particulate pollution which can be calculated in terms of its impact on human health. This level of particulates is governed by factors, relating to the ecosystem, such as the chemistry of the atmosphere, the strength of winds, the stability of soils, and the activity of micro-organisms. When we add ammonia to the ecosystem, the level of particulates changes and we observe an effect on human health. Therefore the effect on human health is a change in the ecosystem service, and should be calculated within the ecosystem service framework.

We would argue that, in reality the treatment of these effects through an ecosystem approach is problematic and may depend on the nature of the ammonia source. In particular, in our analysis we separated point and diffuse sources of ammonia emissions, and this distinction also becomes important in the context of the analysis of air pollution effects on human health.

In the case of point sources, we would argue that there is no terrestrial or aquatic ecosystem involved and that point sources such as intensive animal units can be viewed as 'industrial' sources. Indeed, these large point sources of ammonia are regulated under IPPC just as large power stations are. It could be argued that the atmosphere provides ecosystem services, but this argument is tenuous and would only confuse an analysis based on the ecosystem services as defined by the Millennium Assessment.

However, for diffuse sources of ammonia, the argument of the commentator has more validity, For example, the management of certain ecosystems (i.e. arable land or grassland with manure/slurry spreading) will have marginal effects on ammonia fluxes to the atmosphere, and hence on ammonium aerosols concentrations and human health, as well as ammonium deposition to other sensitive ecosystems. In Table 2.7, for arable and horticultural ecosystem types we have included the relationships between manure/slurry spreading and nitrate leaching and N2O emissions from soil or riparian buffer zones but not with ammonia emissions that affect human health, visibility and climate change (cooling).

In future applications of the Ecosystems Approach for ammonia, it may be appropriate to broaden the analysis to include effects on human health as suggested by the commentator but only for sources which are influenced by ecosystem management and originate from an ecosystem and not from an industrial intensive food production unit.

These considerations would also be important if our analytical framework were to be extended to other air pollutants such as sulphur dioxide or ozone. Almost all sulphur dioxide pollution is from combustion sources and hence emissions from ecosystems do not mediate its impact on human health, climate and the corrosion of materials. The situation is different for ozone which is a secondary pollutant with precursors such as nitrogen oxides, methane and volatile organic compounds that are emitted from a mixture of sources, some of which are anthropogenic (e.g. traffic and oil refineries) and some of which are natural (e.g. biogenic VOCs from vegetation and methane emissions from wetlands). In this case, inclusion of the gaseous emissions within the assessment of ecosystem services would seem to be more appropriate.

In practical terms, the distinction as to whether to include the emissions within the ecosystem services is academic, as the impacts will still need to be reported and considered within the decision making framework.

(ii) Linking ecosystem services to societal benefits

Applying the Millennium Ecosystem Assessment ecosystem services scheme can be problematic as many service categories are inconsistent and tend to overlap and do not clearly distinguish between the capacity of ecosystems to deliver a service and the benefit that people might subsequently derive. In this study this led to the adoption of the 'ecosystem cascade' approach (Haines-Young & Potchin, 2007) to clarify the linkages between biophysical impacts (on air, water, vegetation etc), ecological functions and positive and negative societal outcomes (Section 2.5; Table 2.7). To communicate the principles of an ecosystem services concept in a simplified non-technical way suitable for the decision making process we consider that the Defra Guidelines for valuing ecosystem services could be improved by adopting the ecosystem cascade approach. The stakeholders who commented on this aspect strongly agreed with our approach but urged us to go further, for example:

'*It is in many ways unfortunate that the MEA classification of services has become so widely used as a standard, as it is increasingly emerging that whilst this classification has been valuable in advocacy work, when it comes to analysis and particularly valuation studies the problems of double counting are insurmountable. In the report there is an attempt to use the ecosystems services cascade model of Haines-Young et al. I think a much greater emphasis should have been placed upon this model in the report. As it stands it is difficult to disentangle in the report how the impacts of ammonia on "biophysical structures" (soil water, air and biodiversity) relate to "intermediate services" and final goods and services in the Haines Young et al model. My understanding is that our level of understanding of impacts is highest with regard to biophysical structure and lowest with regard to final goods and services but this does not emerge clearly in the report.*'

'*A particularly refreshing consequence of the recent interest in ecosystem services is the growing emphasis on trying to deliver the economic narrative as the 'crescendo'. The ecological discussions, which I found very informative, were written explicitly for this purpose which is great. That said, I would argue that there is some scope to tighten use of terminology and to more rigorously work through different service typologies and classifications (as recommended in the Fisher et al and Haynes-Young papers). In their defence however they are simply following the guidelines provided by Defra/MA which are difficult to operationalise.*'

As a result of comments received the results of our analysis were modified to specifically consider the biophysical structures impacted, the intermediate services involved and the positive and negative societal outcomes (see Table 2.7). The tabular approach is also recommended by the authors as an alternative to the mapping procedure suggested in the Defra guidelines, especially for applications with complex interactions. Further work on the different service typologies and classifications used is also recommended.

(iii) Complexity and identification of focal areas

The additional complexity introduced by an assessment of air pollution impacts based on ecosystem services, as opposed to more conventional approaches, was clearly identified in our study. Concerns over the higher level of complexity were strongly echoed by the stakeholders consulted:

'*The ecosystem approach however does introduce a much greater level of complexity to analysis of policy and practical solutions. This is inevitable if we are to attempt more holistic analysis. However I think more thought needs to be given to how we make these higher levels of complexity more tractable. I do not anticipate all ecosystem services are equally impacted upon by ammonia. I am sceptical this is significant in the context of coastal defence. However impacts of ammonia on water quality and biodiversity are clearly important and relatively well understood whilst several analyses I have seen all point to the importance in economic terms of carbon sequestration and storage as a very* *economically important ecosystem services in a range of terrestrial and coastal systems. Further discussion and analysis of which are the most important services impacted upon is required if analysis is not to drift into areas of work which although scientifically interesting, from a business perspective are unlikely to be of high economic or other importance*.'

'*The complexity of the approach itself as well as complexity of ecosystems themselves means that by its very nature the report is detailed and needs careful reading and evaluation. There is always a concern that policy makers will seek to simplify things into something that in the end bears little resemblance to the requirements and functioning of the ecosystem itself……you have done an excellent job so far and by your own admission acknowledge that there has been a limited input to the process so far. Well-focused stakeholder participation is essential to inform and engage the approach to develop it and translate it from a conceptual to a practical on the ground delivery'.*

To overcome these issues we recommend that any application of the Ecosystems Approach uses the following steps:

1. Identify clearly the driver(s) of environmental change of interest;
2. Determine the potential marginal changes to ecosystem services (with spatial and temporal dimensions considered) as a result of different policy(s) available to tackle the driver(s);
3. Develop a consensus on the major areas of concern through stakeholder engagement.

Although the stakeholder engagement has been limited in this study the results show that biodiversity, air and water quality and climate change emerge as clearly important areas that are relatively well understood (see Annex III). Having obtained this clear focus and direction from our study, the next task is then to tackle the uncertainties such as lack of scientific and valuation data in the key areas and identify the conflicting demands of the regulatory frameworks involved. These aspects are discussed below.

4.2 Issues related to availability of required scientific and valuation data

(i) Specific data needs

In general, there is a need for information which is specific to the driver(s) of impacts on ecosystem services and of high enough quality and resolution to provide the scientific data and valuations to inform decisions about marginal changes in the provision of services. This study highlights that often the information required to apply the Ecosystems Approach meaningfully is not currently available. Regarding the quantification of impacts to ecological systems and the physical environment the following emerged as key areas where further information is required:

- There is a general lack of knowledge on the effects of land-use change and species composition change on ecosystem services. Most evidence is weak and contradictory, with more reliable information on the relationship between functional types (e.g. trees and N fixers) and ecosystem services rather than on the effects of individual species (Section 2.5; Balvanera et al., 2006). Balvadera et al. (2006) recommend a precautionary approach in the use of management of species changes in general, based on evidence that loss of certain species can be expected to compromise delivery of related ecosystem services. However, the evidence available for this conclusion is based on a limited selection of ecosystem types and further research is necessary in this area. In the case of ammonia, species composition change is one of the major impacts of concern, and it is clear from our study that research is needed to increase the understanding of the relative importance of direct effects of N deposition on ecosystem function compared to those caused indirectly by changes in species composition. Inclusion of the indirect effects will clearly raise the issue of double counting and more generally the issue of how changes in species composition can be more meaningfully placed within the ecosystem services framework, as discussed in Haines-Young & Potchin (2007).

- Quantification is needed of the complex sequence of events that link ammonia emissions to changes in water quality regulation. This would need to include eutrophying and acidifying effects in soils and water bodies, pollution swapping and land management decisions (e.g. use of riparian buffer zones);

- Identifying and quantifying the net effect of changes in ammonia emissions on climate regulation. This would need to include the trade-offs between changes in CO_2, N_2O and CH_4 fluxes from different habitats, interactions between ammonia emissions and secondary aerosol formation (human health impacts and climate 'cooling' effect of ammonium aerosols), and carbon sequestration in vegetation and soils;

In the case of these three key direct and indirect effects on ecosystem services, the timescale of response is an important issue. Changes in emissions and deposition of ammonia (whether increases or decreases) will not have immediate effects on species composition, water quality and climate regulation. There is considerable uncertainty over the timescale of response in different ecosystems, and this will have a significant influence on economic valuation. These issues may be difficult to resolve and the next step may be to conduct a sensitivity analysis to evaluate how different assumptions about timescale affect the predicted changes in services and their valuation.

(ii) Feasibility of required valuation

We conclude that estimating economic and socio-economic values for the changes in ecosystem service delivery following implementation of ammonia control measures is likely to present a considerable challenge for a number of reasons:

- Valuation of the marginal changes in provisioning and regulating services consequent upon the physical changes predicted by the dose-response relationships is difficult because reliable production functions do not appear to exist for some of the services that will be affected (e.g. aquaculture);

- Relevant economic parameters (information on cost of alternatives, price elasticity of demand etc.) are not necessarily readily available for those provisioning and regulating services for which changes in service delivery can be predicted with some accuracy;

- Valuation of ammonia-driven changes in the delivery of recreational and non-market cultural values from aquatic habitats would appear to be easier than for terrestrial habitats at present because of valuation work surrounding the implementation of the Water Framework Directive;

- Valuation in the delivery of cultural and heritage services associated with species restoration are likely to prove particularly difficult because of their extreme sensitivity to context;

- There is a requirement to produce spatially specific valuations for changes in the delivery of many of these services;

- Only valuation functions produced by the most modern valuation methodologies will be appropriate to these tasks, rendering the vast majority of existing studies in valuation collections an unsuitable basis for benefits transfer-derived estimation of the value delivered by ammonia control measures.

We have produced a summary table (Table 4.1) which attempts to describe the reliability and acceptability of each stage of the valuation procedures that are available for each of the key ecosystem services identified in our preliminary analysis for ammonia emission control. A weakest link approach is used where a reliable valuation would be produced for an outcome for which: (a) a reliable dose response function was available with the requisite level of spatial specificity; (b) a well established valuation methodology was available; (c) all the necessary data were available to implement that valuation methodology knowing the 'response' predicted by the dose response function; and (d) appropriate data were available to aggregate site-specific valuations across the whole country. In none of the cases shown in Table 4.1 is overall reliability in the highest class, although it can be seen that in most cases there is the potential to produce a valuation, albeit with varying levels of uncertainty. The relative strictness of our scoring system can be judged against the ‘**’ rating given to the valuation chain currently used in the UK NAQS for reduction in human morbidity and mortality where the weakest link is the uncertainty in the dose-response relationships between NH_3 emission control measures, $PM_{2.5}$ aerosol concentrations and human health outcomes.

Table 4.1 Reliability and acceptability of different stages in the valuation of potential outcomes of changes in NH_3 emissions policies. Key: 0 means missing or not available, through *, ** to ***, where *** means well accepted or generally believed to be reliable. A weakest link approach is used to determine the reliability of the whole valuation chain for the change in good/service/outcome provision concerned. See Technical Annexe IV for details of each valuation approach discussed.

Benefit/disbenefit via particular ecosystem service(s)	**Valuation Methodology**	**Reliability of dose / response function(s)**	**Acceptance of valuation methods**	**Reliability of valuations driven by the 'response'**	**Reliability of summation stage**	**Overall Reliability (following 'weakest link' approach)**
Changes in production of ecosystem goods from aquatic systems (fish)	*Site-specific dose / response followed by changes in producer and consumer surplus via Production function*	*0*	***	*0* *(production functions and elasticity not available)*	*** *would be reliable if relevant dose/response, production function and elasticity data were available*	*0*
Changes in production of ecosystem goods from terrestrial habitats (timber and agriculture)	*Site-specific dose / response followed by changes in producer and consumer surplus via Production function*	**	***	*** for production functions* ** for elasticities (probably)*	*** *would be reliable if relevant dose/response, production function and elasticity data were available*	**
Drinking water provision	*Site-specific dose / response followed by changes in producer and consumer surplus via Production function* ***or*** *site-specific dose / response followed by direct WTP methods*	**	***	*0* *dose /response relationships are available, but production functions are commercially sensitive* ** Direct WTP studies lacking for the UK*	*** *would be reliable if relevant dose/response, production function and elasticity data were available* *or* * *only most recent direct WTP estimations suitable for benefits transfer and thus summation*	*0 for production function approaches* ** for direct WTP estimation*
Water-based recreation & tourism	*Site-specific dose / response followed by Direct and indirect WTP methods*	**	***	*** (at best)* *Recent valuations of recreational benefits of improved water quality for the WFD provide a good start here*	** *only most recent direct and indirect WTP estimations suitable for benefits transfer and thus summation*	**

Benefit/disbenefit via particular ecosystem service(s)	**Valuation Methodology**	**Reliability of dose / response function(s)**	**Acceptance of valuation methods**	**Reliability of valuations driven by the 'response'**	**Reliability of summation stage**	**Overall Reliability (following 'weakest link' approach)**
Land-based recreation and tourism	*Site-specific dose / response followed by Direct and indirect WTP methods*	**	***	* *(at best)* *Few existing recreational valuations assess WTP for changes in species diversity which are likely to result from NH_3 policy changes.*	** *only most recent direct and indirect WTP estimations suitable for benefits transfer and thus summation*	*
Changes in availability of biochemical/genetic resources from aquatic and terrestrial species	*Site-specific dose / response followed by estimation of Option value & Quasi-option value via direct WTP methods*	**	*	*?* *unknown*	*0* *None available*	*0*
Changes in climate change regulation (via carbon sequestration of vegetation and soils and change in emission of GHGs and/or potential for pollution swapping)	*Site-specific dose/response relationships followed by valuation of changes in GHG emissions via shadow price of carbon (SPC) method*	*	** *Considerable controversy surrounds specification of the 'correct' SPC*	*** *SPC approach follows a prescribed methodology once a change in GHG emissions has been established*	*** *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 NH_3 measures, $PM_{2.5}$ aerosol concentrations and human health outcomes. Changes in health outcomes valued via QALY and VOSL methods	**	*** Well established methodology	*** QALY and VOSL approaches follow a prescribed methodology once the QALY consequences of changes in HN_3 / $PM_{2.5}$ aerosol concentrations has been established	*** *Site-specific dose response followed by summation to estimate effects on whole population; no need for benefits transfer*	**

(iii) Is the information available suitable for decision making?

The stakeholders consulted raised some interesting points regarding the true costs and benefits of ammonia mitigation, for example:

'*As with the vast majority of environmental issues, there is usually an inherent tension between 'marketed goods' (i.e. products from agriculture that are bought and sold) and 'non-marketed goods' (i.e. many ecosystem services which are not but nevertheless impact on people's welfare in some way). For me the paper usefully identified the potential trade-offs on the benefits side - between different services & what I think it called 'pollution shifting' - but didn't reflect at all on the potential costs of the 'abatement measures' considered. At the end of the day, some assessment of costs and benefits is needed. Perhaps this was beyond the scope of the study but I was surprised by its absence and left without a sense of whether, in economic terms, any of the abatement measures they proposed are worth it*.'

'*In the assessment of impacts, the report concedes significant uncertainties and suggests that accounting for these uncertainties would require an extensive GIS analyses based on detailed values of NHx deposition due to different sources. I would agree with this view but would suggest that we are currently lacking in the detailed information on farm practices (livestock production, manure management etc) that we would need to inform such assessments. This has been a general concern of ours over a long time, but I feel that it is particularly important for your assessments. Such information is also lacking in terms of a robust assessment of the true costs of policy implementation (whatever the policy may be). How can such costs be credibly evaluated without a detailed knowledge of farming practices and to what extent any particular measure can or should be implemented? This is a real potential barrier to implementation of any significant policy change*.'

'*It is agreed that a more detailed evaluation of the 'pollution swapping' aspects would also be essential, e.g. to assess whether reducing ammonia emissions by low emissions spreading techniques leads to an overall increase in N_2O emissions. Conflicts between reducing emissions in one system and increasing them elsewhere are highlighted. One aspect which does not appear to have been considered sufficiently (at all?) is the high price of energy & fertiliser - to such an extent in the past year that manures are now worth per unit mass, some 4x the potential financial value of some 18 months ago. Thus, whilst in the past ammonia mitigation measures for manure applications may have been applied to comply with regulation, the tendency would have tended still to ignore the increased retained N value - thus there may be knock-on impacts in terms of reduced NO_3 leaching or N_2O emissions?*'

'*In terms of the approach to valuation, the paper is spot on in focussing on 'marginal' changes in service provision. What concerns me slightly about the ES approach more generally is that potentially significant ecological changes can lead to relatively minor changes in service provision and that as we move further down the ES cascade towards the economics end, the ripples of any change get weaker and weaker. Part of this is because we can not 'value' everything we wish due to data limitations at all points along the cascade (as noted in the paper) but some of it relates more fundamentally to WHAT people actually value. E.g. do people really value species rich grassland significantly more than average grassland? As a result, the study draws a lot of blanks and falls back on a) the recreational benefits of improved water quality which is where most work has been done in the past, and b) replacement cost approaches around water treatment.*'

These stakeholder comments prompted us to consider in more detail, the potential for developing a cost-benefit analysis, based on the analytical framework that we have developed in this project. Clearly, the outcome of a detailed quantitative analysis of specific national scenarios in terms of all national ecosystem services is beyond the scope of the present exercise. Indeed, it could be argued that it may never be feasible. Furthermore, the present report is too preliminary to provide any meaningful answer to the question of whether, in economic terms, any of the abatement measures proposed have economic benefits that outweigh the costs. Nevertheless, if the UK government signs up to a national emissions ceiling to be achieved by 2020, it seems clear from our analysis that is possible to ask the question: what is the optimal combination of measures which should be adopted to achieve this goal? The analysis conducted for Defra to date has focussed on the costs and feasibility of different measures, and their implications in terms of pollution swapping (Misselbrook, 2007). It seems perfectly feasible to us to extend this analysis to the positive and negative effects on ecosystem services, using some of the approaches and information that we have identified.

To illustrate what a cost-benefit analysis could look like, we have taken the estimated cost of the proposed abatement scenarios used in this study (Misselbrook, 2007) and compared them with the Defra figures for the health impact of ammonia emissions and our very preliminary figures for marginal changes to carbon sequestration by vegetation and GHG emission from soils (Section 2.7.2; Table 4.2). The cost/benefit is calculated by multiplying the unit pollution cost/benefit (million £ per tonne of ammonia reduced) by the amount of ammonia that a given scenario reduces ammonia emissions by relative to the existing NEC 2010 target of 297 kt of ammonia. It must be stressed that this is only a partial analysis and to be realistic all the positive and negative effects on all the ecosystem services identified would have to be considered. Table 4.2 shows that health benefits alone cannot justify the cost of the scenarios proposed and that there is definite potential for ecosystem services to make a difference to the outcome of the analysis given the magnitude of our estimates and the fact that many of the ecosystem service effects identified in this study have positive outcomes (see Table 2.7). In this preliminary analysis, our estimated economic costs and benefits of climate regulation from changes in carbon sequestration and soil N_2O and other fluxes approximately balance, and hence their overall impact on the cost/benefit comparison is not great. However, the uncertainties are large, they may be influenced differentially by specific control measures, and there are many other ecosystem services which need to be considered.

Table 4.2 Illustrative cost/benefit analysis comparing cost (Million £) of suggested ammonia reduction scenarios (Misselbrook, 2007) and the cost/benefit (Million £) arising from a limited selection of marginal changes to impacts (estimated £ per tonne NH_3 reduced: health, + £1,407; carbon sequestration by vegetation - £489; and GHG emission from soils, + £411).

	Health benefit related to reduced $PM_{2.5}$ (Million £)	Carbon sequestration vegetation (Million £)	GHG benefit from soils (Million £)	Cost of Scenario[2] (Million £)
Scenario 1 (~47kt reduction[1])	66	-23	19	-140
Scenario 2 (~67kt reduction[1])	94	-33	28	-189
Scenario 3 (~87kt reduction[1])	122	-43	36	-306

[1]compared to 2010 NEC target of 297kt
[2] According to Misselbrook, 2007

Two key practical questions for implementation would then be how well could the analysis of specific measures be developed to assess policy-specific impacts on climate regulation and how far could it be expanded to include a greater range of specific effects on ecosystem services which could be related to costs of implementation and other benefits (e.g. those on human health from reduced concentrations of ammonium sulphate)? If this could be done, it might significantly alter the choice of specific measures that are selected to achieve target emission ceilings.

This leads to the general question of how likely is it that the relative costs and benefits of different impacts of ammonia can be valued accurately enough, both as totals and per unit of N emission, that they can be used to meaningfully inform the prioritization of abatement measures? For example, should equal effort be put into water, air quality, greenhouse gas balance, ecosystems and biodiversity and soils or are there arguments that provide good reasons for prioritizing certain areas? As we have shown in Table 4.1 there is potential to produce reliable valuations in most areas in the future but currently there are still some substantial gaps in the methodologies and data availability. In addition to this, the issue raised by a stakeholder of obtaining detailed information on farm practices is also crucial as the way that certain abatement methods are implemented, and even the weather conditions prevalent at the time, can have significant bearing on the results and the potential for unwanted pollution swapping events. So, for ammonia, improved techniques may also be required to obtain reliable estimates of the true costs of abatement measures at the farm level.

However, further development of such a cost/benefits analysis of specific emission control measures can, at least in the neat future, only be based on those ecosystem services than can be readily valued. This highlights the point made by one stakeholder, that there is a danger that the value that society puts on a particular ecological effect, such as loss of certain species, may underestimate the true significance of the ecological change that results. There is a at least a potential danger that implementing a partial cost-benefit analysis based for example on ecosystem services such as water quality and climate regulation could lead to a policy which could lead to adverse effects on other services where data are lacking. In such cases the qualitative assessment of the relative importance of the different effects on ecosystem services becomes even more crucial.

(iii) Usefulness of the qualitative stage

Despite the absence of a rigorous quantitative analysis all stages of the ecosystem cascade, we would also argue that application of the ecosystem approach still has many benefits. The stakeholders generally agreed, arguing that the lack of the detailed data required does not necessarily rule the out ecosystem approach as a useful policy tool. For example:

‘*Whether or not valuation is possible, surely the approach provides a framework for at least a qualitative assessment of the wider benefits/disbenefits of a policy? This is likely to become increasingly important as policies focus on climate change mitigation (and adaptation). Even a limited application of the ES approach through a GIS would help to target holistic actions, and not just AQ standards – i.e. WFD, NVZ, climate change, renewable energy and AQ).’*

‘*The principal weakness is that data are available for but few of the ecosystem services. This is likely to lead to debate over the fine detail of any assessment rather than the general direction. For this reason, I suggest that, in the first instance, a qualitative interpretation of ecosystem services is more appropriate to policy development than a formal cost benefit analysis’*

It is clear from our study that the qualitative stage of the Ecosystems Approach analysis using the impact pathway approach is the stage where a truly holistic appraisal of the suggested policy change is made. The quality of choices made at this stage is then determined by the knowledge and appropriateness of the stakeholders that are consulted. It is also at this stage that balance can be sought between the many conflicting considerations that need to be made, regarding ecological, financial, cultural and even spiritual significance. These points were also made clearly by the stakeholders consulted, most of whom are involved in the types of organisations that would be called upon to implement the Ecosystems Approach. In summary, although this section reveals many uncertainties and inadequacies concerning the various different types of information available for implementing an Ecosystems Approach for ammonia, as a strong conclusion from this study is that the qualitative stage of the impact pathway approach is potentially a very useful tool to aid integrated policy development.

4.3 Issues related to policy application

(i) Treatment of effect of air quality on human health

We have separated analysis of effects of ammonia emissions from agricultural practices on human health which are considered in national air quality management from effects of deposition on ecosystem services which are the focus of this report. Ammonium aerosol formation from agricultural emissions that can affect climate and visibility are also considered not to involve ecosystem services.

However, on commentator has argued:

‘*The report considers that the effect of ammonia, for example, on carbon sequestration can be calculated as a change in delivery of the ecosystem service of carbon sequestration i.e. the ecosystem provides some level of sequestration in the absence of ammonia pollution, and a different amount of sequestration when ammonia is present. We can then calculate the value of this change in ecosystem service. By analogy, the ecosystem provides a certain level of particulate pollution which can be calculated in terms of its impact on human health. This level of particulates is governed by factors such as the chemistry of the atmosphere, the strength of winds, the stability of soils, and the activity of micro-organisms. When we add ammonia to the ecosystem, the level of particulates changes and we observe an effect on human health.*

Therefore the effect on human health is a change in the ecosystem service, and should be calculated within the ecosystem service framework.

I can see that in other policy areas there may be cases where there are effects of human activity which are not mediated by ecosystems – an example would be the effect of road transport. Cars affect the ecosystem through pollution, but they are also involved in accidents which kill and injure people directly.’

[Note to commentator: NOx from vehicles also contributes to human mortality and morbidity directly without any ecosystem involvement.]

In reality the treatment of these effects through an ecosystem approach is problematic and may depend on the nature of the ammonia source. In particular, we separated in our analysis point sources and diffuse sources and we will consider threes in turn:

Point sources: No ecosystem involved and intensive animal units can be viewed as ‘industrial’ sources and indeed they are regulated under IPPC just as large power stations are.

Diffuse sources: It could be argued that management of certain ecosystems (i.e. arable land or grassland with manure/slurry spreading) will have marginal effects on ammonia flux and should therefore be included in an ecosystems approach. However, in our study we keep the distinction of considering only the deposition that results from ammonia emissions affects ecosystem services.

Therefore, decisions on how effects on air quality are integrated into an Ecosystems Approach for ammonia depend on the nature of the source under consideration.

This would also be true if our analysis were to be extended to other air pollutants such as sulphur dioxide or ozone. All sulphur dioxide pollution is from combustion sources and ecosystems do not mediate its impact on human health, climate and the corrosion of materials. Ozone is a secondary pollutant with precursors such as nitrogen oxides, methane and volatile organic compounds that are emitted from a mixture of sources, only some of which involve ecosystem services.

(i) **Regulatory Frameworks**

How readily could the framework that we have proposed for the application of the Ecosystems Approach for ammonia be applied within existing regulatory frameworks? If the intention is to apply the holistic analysis of costs and benefits at the scale of national analysis of the costs and benefits of different policy options, then the existing framework of air quality management is already fit for purpose. The Ecosystems Approach would provide a more transparent assessment of all the consequences for ecosystem services of particular policy options and a basis for extending valuation to a large range of services than are currently considered. If this analysis was accompanied by a transparent evaluation of the quality of the underlying data, such as that provided in Table 4.1, the limitations of any valuation would be clear. This evaluation applies to ammonia, but there is no stage of our analysis that could not in principle equally be applied to other air pollutants, or indeed a wider range of air, soil and water contaminants. However, more detailed evaluation is needed to verify that this is the case, and to identify any constraints to national policy application, apart from the lack of adequate data.

However, existing regulatory frameworks in the UK may not be adequate to address a more detail spatial or habitat specific application of the Ecosystems Approach for ammonia. Holistic assessment introduces the potential for the priorities of different policy areas, such as conservation of plant species, maintenance of water and air quality and issues of climate change regulation, to clash. This is of course not new; tensions between different policy objectives are inherent in many regulatory issues, and a clear benefit of the Ecosystems Approach is that these are made explicit in the analysis. Nevertheless, some key practical questions for implementation remain, including:-

- Since the effects of ammonia emissions on ecosystem services differ between habitats, and sources and habitats have interrelated spatial distributions, different measures might be applied to different degrees in different parts of the country to optimise the outcome for ecosystem services. This would mean that there was not a ‘level playing field’ between farmers and other producers of ammonia emissions, since costs for one producer would change differently from those of another. While this arises already in regulation of the farming industry, the current limitations of the quantification of effects on ecosystem services mean that the evidence base to support regulation leading to differential costs for producers is weak, and may be open to legal and other challenges.

- The application of the ecosystem services approach would need to considered in the context of other legislation e.g. some species are actually protected under Natura 2000 and there is a potential for conflict between maximizing ecosystem services for climate protection and conservation priorities.

- The uncertainties involved may mean that policy instruments may require a precautionary tone, which may be problematic for certain stakeholders.

The stakeholders consulted also commented, as in the examples below, on the policy regulation issue, including the concern reflected in this report over the compatibility of critical loads and levels with the Ecosystems Approach:

‘*So far there has been no analysis of the indirect contribution of abatement measures to other ecosystem services – for example, tree planting to intercept emissions and protect water courses will contribute to ‘climate regulation’. Such double impacts are important in policy terms – particularly the focus of the climate change bill on sustainable development.*’

‘*Current approaches look at air quality objectives in isolation and adopt a precautionary approach. This may be at odds with other policy objectives*.’

‘*Critical loads were developed to protect specific functions or indicators (for example, the CL for conifer woodland was developed to protect the potential to revert to semi-natural land cover, rather than anything else). The precautionary approach was applied through ‘potential for change’. The CLs were set on the basis of the lowest* *concentration where effects of any type were observed. This is not necessarily true for other ecosystem services. For example, increased herbivory was cited as an impact of enhanced nitrogen deposition – because the trees grew faster and they became visible above the grass sward…. Effectively, the CL approach is designed to preserve habitats created by management over millenia because of the nature of the UK landscape; it is not designed to protect natural ecosystems.*'

The key area of conflict identified here is between nature conservation and climate regulation, as the systems that are protected in this country are largely semi-natural systems created by particular forms of human management, whereas carbon storage and the net benefit for climate regulation may be enhanced by allowing succession to primary forest systems. This is relevant for ammonia because increased nitrogen inputs may enhance carbon storage but damage protected semi-natural ecosystems. It is arguable whether the ecosystem services related to recreation, education, cultural values etc. provided by say a lowland heath and the forest to which it might be transformed without management intervention are greater, but the objectives of current conservation policy are clear. As the stakeholder comments reveal, this is an important issue for reconciling the objectives of air quality management in terms of both critical loads and ecosystem services, for ammonia and other pollutants, but it has wider implications for implementation of regulatory approaches based on ecosystem services. Detailed analysis of this issue is outside the scope of this project, but as UK policy becomes more strongly focussed on management for climate regulation, it is an important one for further analysis.

In conclusion, considering our analysis and the comments of various stakeholders we would identify the following barriers to implementation of the Ecosystems Approach in the UK which have emerged from our study of ammonia, but have wider relevance:-

- Development of an ecosystems cascade approach is essential to improve the current guidelines to operationalise the Ecosystems Approach;
- Development is needed of methods of dealing with the high degree of complexity introduced by application of the Ecosystems Approach, and identifying objectively the key areas, and key impacts on services, which need detailed analysis
- The lack of available data quantifying the value of marginal changes in the delivery of individual ecosystem services is a major barrier which can only be overcome by investment in new data collection;
- Analysis to identify the key gaps in data which prevent reliable valuation for a particular issue is possible using a 'weakest link' approach;
- Conflicts between different policy objectives can be made transparent by application of the Ecosystems Approach, but, especially given the current limitations in valuation and the potential for low national valuation of services that are highly valued by certain groups; a framework is needed for how the Ecosystems Approach is to be used in resolving these conflicts.

We have shown in this report that the Ecosystems Approach has the potential in principle for successful application to the issues of national policy on control of ammonia emissions, and can provide important new information that is not provided by current methods of policy analysis. We consider that this conclusion is likely to apply to other areas of national air quality management, and more broadly in the national policy on environmental contaminants. The main barriers relate to the analytical frameworks for its application, and the data available for detailed analysis. Both of these barriers can be overcome by further research, although the amount for work needed to resolve gaps in data is substantial. Unless this is done, the results of any detailed application of the Ecosystems Approach may be so uncertain that they will lack the robustness and reliability which is needed to resolve conflicts between policy outcomes.

5. Requirements for implementation of the Ecosystems Approach for ammonia in the short and long-term (Objective 4)

5.1 Challenges to implementing the Ecosystems Approach

Whilst the Ecosystems Approach is accepted by many within the policy and academic community as a progressive framework for a full costing of all the environmental and social effects of management decisions, implementing the Ecosystems Approach in practise presents a number of challenges.

1. Public understanding and acceptance of the Ecosystems Approach
The Ecosystems Approach is not yet well understood or appreciated by the majority of those involved in management decisions. In the present context, the most relevant stakeholders are those in the farming industry, but few of these are engaged with the concept and most are not aware of it. Successful implementation of an Ecosystems Approach within the farming community will require extensive consultation and demonstration and it is likely to take many years before it becomes part of the culture of that industry. There will need to be an extensive period of promotion and demonstration of the value of an Ecosystems Approach to all of those involved, in all stages and levels of environmental management and planning, to ensure its acceptance as a management tool.

There is a large amount of variation in the level of awareness and acceptance of the need for an Ecosystems Approach in the UK community of practitioners that are involved in its implementation. If the Ecosystems Approach is to be successfully implemented then there is a need for a shift in culture across all stakeholders, including farmers, agencies and regulators. Improved information on the link between the provision of services and the benefits to society will go a long way to addressing this problem.

The stakeholder consultation for this report only concentrated on key individuals in agencies such as Natural England, ADAS, the JNCC and the Forestry Commission. Comments received from these individuals were generally very supportive of the ecosystem services framework proposed in the report, for example:-:

'*I see this as a positive development and, given that the current measures and UK target scenarios appear unlikely to provide significant benefits, a new approach which does not fall into the "one size fits all" category, needs to be given serious consideration. If such an approach would allow a regional/local implementation of selective strategies which might be supported in some way (e.g. by regional development agency grant?) then farmers might take more "ownership" of the issue - why should they implement more & more costly measures on their farms, which contribute nothing to their bottom line profitability, when they are told anyway that the measures are inadequate and the results insufficient.*'

'*We are very supportive of the general approach which highlights the much wider range of benefits which might be derived form mitigating the effects of ammonia deposition. The concept of pollution/ ES trade-offs is introduced here and is a key issue which has not been much examined elsewhere so far*.'

'Such an approach is fully in line with England's Forestry Strategy…We see the need for a strategic resource to deliver the full range of ecosystem services that trees and woodlands can provide ….I would argue that in terms of grant aid (not regulation) the English Woodland Grant Scheme (EWGS) does provide a more holistic vision; also the EIA (forestry) regulations allow the various ecosystem services to be explored – albeit without defining them explicitly. The current development of open habitats policy by FC England is also an expression of ecosystem services in action. There is no need for the issues to clash, just for their relative merits to be explored (often qualitatively).'

However, a much wider consultation with all relevant stakeholders is clearly essential to build the wider consensus that would be needed to take the approach forward.

2. Skills and knowledge gaps
The Ecosystems Approach assumes inter-disciplinary collaborations between the natural, economic and social sciences which have historically proved difficult to achieve, but which show signs of improving in recent years. Nevertheless, more effort is needed to develop incentives and mechanisms for allowing interdisciplinary working. In addition, the skill-base within these disciplines, particularly in the areas of environmental economics and ecological economics is presently limited within the UK and may prove a limiting factor for practical implementation of the Ecosystems Approach. Investment in training and education in these areas will be required in order to create the

manpower needed to deliver the Ecosystems Approach. In addition, there are significant knowledge gaps in our understanding of the relationships between biophysical factors, such as biodiversity, and the delivery of ecosystem services, as well as the feedbacks and interactions between such factors which will arise when ecosystems are managed for multiple services. Much basic research is needed to better understand these interactions and trade-offs. The UK research councils are beginning to invest in these areas.

General stakeholder comments covering knowledge gaps included the following:

'*It would be valuable to carry out similar analyses of other policy areas to see if general themes emerge.*'

'*The potential complexity of robustly applying an ecosystems services approach is striking but I guess inevitable if we are to take proper account of all impacts.*'

'I think a case study or two in an area like East Anglia close to sensitive habitats would provide a demonstrable example of an ecosystems based approach that links all the drivers such as Water Framework Directive, Nitrates Directive, Habitats Directive etc but links it with measurables such as water and air quality and also demonstrable ecological conservation based objectives.'

The discussion in the Section 4.3 highlighted the fact that despite the technical and data availability problems the qualitative analysis stage of the Ecosystems Approach has considerable potential as a useful tool to aid integrated policy development. It also seems likely that the general advantages, barriers and challenges of using the Ecosystems Approach identified in this study will be common to other policy areas.

3. Regulatory frameworks and management for multiple services

One of the major benefits of an Ecosystems Approach is that it allows an appreciation of all the effects of management decisions on the full range of services provided by that system. Managing an area to increase the delivery of one service, such as food production, may have impacts on other provisioning services, as well as a range of regulating and cultural services. These impacts can be fully costed and compared against the benefits of the planned management intervention to allow a decision on whether that management should be permitted.

Such management must operate within the constraints of existing regulations and policies. However, the holistic nature of the Ecosystems Approach will not be easy to relate to much of the existing regulatory framework, which is designed mainly for the protection of specific elements related to individual services. For instance, the increase in ammonia emissions from intensive farming operations may have negative impacts on plant species richness, on human health, on recreation and tourism and on soil and freshwater quality, each of which is covered by a different regulatory framework. More holistic frameworks are needed to simplify the regulatory environment within which management can operate using the Ecosystems Approach. The Water Framework Directive provides an exemplar of a more holistic framework within which an Ecosystems Approach can operate (e.g. Hartje and Klaphake, 2003), but it is unlikely to prove entirely comprehensive across all services.

The stakeholders consulted also mentioned these important areas, for instance:-

'*I have concerns that cultural expectations will drive ecological expectations and from my experiences across Europe there are a wide range of environmental outcomes proffered across various member states. The benefit of an ecosystem approach is that it brings together synergistic and also conflicting drivers. Cultural expectations may however rank these in different orders depending on the ecological outcome sought. However, at this stage if we focus on simpler outcomes such as the provision of "clean" air soil and water it will be much easier to provide a datum to provide a quantitative measurable outcome. This is a key message contained throughout the report and the ecological uncertainties (at this stage) need to be continually highlighted. I think that there is a huge amount of legislation that is poorly or simply not applied which could tackle a number of our current concerns. An integration of existing regulatory drivers could form a framework on which to superimpose an ecosystem approach.'*

'Adoption of an ecosystem services approach – at least in a qualitative format in the first instance - is necessary. As the report states, this has been achieved to some extent through the WFD and associated River Basin Management Plans. However, WFD has not necessarily made the necessary links with other policies that could have been achieved – although I anticipate that this may follow as the measures to achieve good status are developed. Such an approach will become increasingly important as measures to address climate change (mitigation and adaptation) are developed as a result of the Climate Change Bill; this requires that the principles of sustainable development are followed for all measures, thus reflecting the ecosystem services approach.'

'*Generally the problems we face relate to local delivery that is i. lack of methodologies which allow us to interpret which services are or should be delivered at given localities, ii. understanding of what impedes or threatens services locally, iii. having methods of land and water management available which address any problems and iv. having not just regulation but also the incentives to ensure ecosystem services are delivered.'*

6. Main Conclusions

The main conclusions of the report are summarised here in terms of the original four objectives.

Objective 1: To test the potential for an Ecosystems Approach to make a meaningful holistic assessment of the impacts on ecosystem services of different abatement options for ammonia emissions to the air at national level in the UK.

Main Results

This report shows that it is feasible to embed an ecosystem services framework for air quality policy on agricultural ammonia emissions within an Ecosystems Approach in the UK. The main results of the study are:

- The most important, and relatively well understood, positive changes to ecosystem services resulting from reduced ammonia emissions were related to air and water quality, species composition and climate regulation (i.e. decrease in greenhouse gas (GHG) emissions from soils). Important negative changes occurred where the fertilizing effect of nitrogen deposition had had a beneficial effect on harvested goods and carbon sequestration by vegetation and where changes in methods of slurry storage and application may lead to decreased ammonia emission at the expense of increased GHG emissions or nitrate leaching from soils (i.e. pollution swapping).
- Wetlands, broadleaf woodlands and grasslands were predicted to show the greatest change in ecosystem services due to their large geographical overlap with ammonia sources in the UK.
- Preliminary calculations, for changes in climate regulation only, show that the scale of benefits and costs is likely to be of a similar order of magnitude to that estimated for the human health impacts of ammonia. The analysis also suggests that the costs of the more stringent ammonia emission reduction being considered for 2020 are unlikely to be matched by savings from health benefits alone and that including benefits related to ecosystems services could potentially alter this balance, especially given the magnitude of the figures estimated and the fact that many of the effects on ecosystem services are positive (providing societal benefits).

Objective 2: To compare the Ecosystems Approach with current conventional air quality management approaches to determine the relative strengths and weaknesses of each approach.

The strengths that this study has identified for the application of the Ecosystems Approach to air quality management for ammonia can be related to Defra's core principles (Defra, 2007b) in the following way:

- It provides a framework for systematic assessment of ecological effects; it allows information on specific species, process and habitats to be synthesised;
- It captures important regulating effects and can highlight pollution swapping potential, such as those for water quality and climate regulation, which are not currently considered in the National Air Quality Strategy;
- It can be adapted to include consideration of effects at various scales and at a national level;
- It can support the definition and application of environmental limits by defining the specific ecosystem services which they can protect;
- It provides a framework for a comprehensive holistic assessment, involving human health, ecosystem services and the built environment;
- It provides a framework for adaptive management, since the implications for different ecosystem services of alternative emission control policies, and the possible tradeoffs involved, can be clearly identified.

The comparison with existing air quality management also showed that the use of the ecosystem services framework would clearly allow more explicit consideration of the consequences of critical load and level exceedance, and hence would add a new dimension to policy evaluation. The weaknesses of the Ecosystems Approach that were identified mainly relate to its generic application, rather than to the specific context of air quality management.

The existing regulatory frameworks for air quality in the UK do not adequately address the spatial and habitat specific issues that emerge from the application of the Ecosystems Approach for ammonia. The major advantage of the Ecosystems Approach is that it captures a host of effects of ammonia emissions that are currently not covered by the NAQS e.g. point source effects, pollution swapping, water quality and regulation, carbon sequestration and ecosystem goods and cultural aspects. However, the existing NAQS does have the advantage that, in comparison with the Ecosystems Approach, it is relatively straightforward to apply and provides a 'level playing field' for stakeholders affected by government policy across the country.

If a spatial strategy for ammonia was based on ecosystem services rather than critical load exceedance, the priorities in terms of both the location and techniques used for emissions control could be quite different because :-

- There are potential benefits of ammonia emissions on ecosystem services below a particular critical load or ecological threshold;
- The point at which adverse effects on ecosystem services dominate any positive effects, for example, on production or carbon uptake, is habitat-specific, and indeed even site specific.

Objective 3: To identify potential barriers, and their causes, to the implementation of an Ecosystems Approach for air quality policy development at national level in the UK.

This assessment considered specific (ammonia related) and general barriers to implementing an Ecosystems Approach and it also benefited from comments on an interim report from a small group of air quality management experts and stakeholders. The following are the salient points:

- Research is needed in certain key areas to overcome limitations of available knowledge on the benefits of reduced ammonia emission. These include:- quantification of carbon sequestration in vegetation and soils; net nitrous oxide emissions from agricultural land and other habitats; the complex linkages between ammonia emissions and water quality and regulation; the relative importance of direct effects of nitrogen deposition on ecosystem function compared to those caused indirectly by changes in species composition; and improved valuation of marginal changes related to reduced ammonia emissions;
- Only economic valuation functions produced by the most modern methodologies will be appropriate to accurately value the socio-economic benefits of implementing ammonia control; the vast majority of existing studies in valuation provide an unsuitable basis for benefits transfer-derived estimation of the value delivered by ammonia control measures.
- There is a need to communicate the ecosystem services concept in a non-technical way suitable for the decision making process - clarifying links between ecosystem services and the positive and negative outcomes for society using the ‘ecosystems cascade approach’ is one way that this could be achieved;
- Our analysis shows that application of the Ecosystems Approach can highlight conflicts between the priorities of existing policy areas, such as conservation of plant species, maintenance of water and air quality and issues of climate change regulation; However, the extent to which the Ecosystems Approach could be used to resolve such conflicts is very uncertain,
- The large uncertainties involved in applying the Ecosystems Approach may mean that policy instruments may require a precautionary tone, which may be problematic for certain stakeholders.

Objective 4: To consider the type of short and long-term changes to the policy /regulatory framework, and research and development needs, that may be needed to address barriers identified in 3.

Short-term

Studies need to be taken to fill the skill and information gaps, as well as initiatives to promote the interdisciplinary work required for the successive implementation of the Ecosystems Approach.

Studies are also required on the linkages and contrasting requirements of the various regulatory frameworks e.g. a challenge will be to use the Ecosystems Approach effectively to link catchment-specific approaches such as the WFD with the aim of meeting national emissions targets. This implies addressing the challenge of the spatial distribution of measures to reduce ammonia emissions that would maximise the benefits for ecosystem services.

A full cost-benefit analysis is still a long way off as valuing changes in the delivery of ecosystem services arising from changes in ammonia emission management policies is challenging. This conceded, the qualitative framework with a spatial dimension already holds much promise as a holistic policy tool, especially for the identification of priority areas, pollution swapping elements and the trade-offs associated with any particular policy.

Long-term

This study has shown that the regulatory agencies and research community are very receptive to the concept of an Ecosystems Approach. However, the long-term challenge faced is gaining public understanding and acceptance of the Ecosystems Approach and in particular gaining support from the farming community and other ammonia producers for the approach.

Creating the necessary regulatory frameworks and management practices for multiple services is a major long-term challenge.

Finally, two potential useful approaches have emerged from this study that could be developed in the longer term:

(i) Valuing the benefits for ecological services of ammonia emission reduction could potentially be based on exposure-response relationships with no threshold. Such an approach would be analogous to the approach used to value the health benefits of ammonia emission control, through the contribution to secondary aerosol formation, and hence would provide a consistent methodological framework for an integrated analysis by application of the Ecosystems Approach. The potential is emerging for an integrated ecosystem approach to valuing the benefits of reduced ammonia emissions, based on exposure-response relationships and dynamic models, which could replace the current approach based on environmental thresholds.

(ii) The difficulty in valuing ecosystem effects may not be the level of assumption that is needed (which may be no greater those used to value benefits for human health), but the lack of standardised and agreed methods to do so. Assessment of the benefits for human health relies on putting values on a human life, or on ill health, through standard measures. The potential value of the Ecosystems Approach is that it allows a wide range of ecological effects to be considered in terms of a narrower suite of ecosystem services – i.e. it creates a limited range of outcomes which can in principle be valued.

7. Future Work

It is clear from the preceding sections that there are major generic challenges to implementation of an Ecosystems Approach to any policy area. Here we focus only on specific needs for further work to improve the basis for application of an Ecosystems Approach to the issue of agricultural ammonia emissions.

Recommendations

- Development is needed of methods of dealing with the high degree of complexity introduced by application of the Ecosystems Approach,
- If detailed application of the Ecosystems Approach is limited to a few key ecosystem services, methods of identifying objectively the key areas need detailed development and appropriate stakeholder involvement;
- The lack of available data quantifying the value of marginal changes in the delivery of individual ecosystem services is a major barrier which can only be overcome by investment in new data collection, especially considering the true costs associated with farming practices; Analysis to identify the key gaps in data which prevent reliable valuation for a particular issue is possible using a ‘weakest link’ approach;
- This study clearly reveals many uncertainties and inadequacies concerning the information available for implementing an Ecosystems Approach for ammonia, but a strong conclusion from this study is that the qualitative stage of the impact pathway approach is already a very useful tool to aid integrated policy development;
- Conflicts between different policy objectives can be made transparent by application of the Ecosystems Approach, but, especially given the current limitations in valuation and the potential for low national valuation of services that are highly valued by certain groups, a framework is needed for how the Ecosystems Approach could be used for resolving these conflicts;
- Case studies with stakeholder involvement in areas close to sensitive habitats would provide demonstrable examples of the ecosystems based approach and could also allow links to other policy drivers to be made more explicit.

8. Communication of results

This work is highly relevant to ongoing discussions in the EU and within CLRTAP on ammonia emission control. Kevin Hicks attended the inaugural meeting of the new Task Force on Reactive Nitrogen (TFRN) 20-22 May 2008 Wageningen (Netherlands). He gave a talk to the TFRN entitled ‘Considering an Ecosystems Approach as a way forward for the regulation of ammonia emissions in the UK'. Delegates from Finland, France and the USA were interested to hear about the results of the project as they are also interested in applying similar approaches and the final report will be circulated to them. Whilst in Wageningen Kevin Hicks also took part in the European Science Foundations (ESF) drafting workshop of the Nitrogen in Europe (NINE) European Nitrogen Assessment (ENA) for the chapters on Multiple Nitrogen Problems and Current Policy Measures.

As a direct result of his attendance of these meetings Kevin Hicks was subsequently invited by Hans van Grinsven, of the Netherlands Environmental Assessment Agency (MNP), to co-organise a further NINE ENA meeting on drafting the chapter on costing nitrogen in the environment. This drafting workshop will take place in York 2-4 December 2008 and will use the results and contacts made in the present project extensively. Defra will also be invited to attend and it is hoped that an economist from the present project's steering group will be able to contribute to the meeting from a policy maker perspective.

The results of the project once agreed with Defra will be used to inform discussions within Defra and with other UK experts and stakeholders. As the project forms part of a wider initiative on ecosystem services by Defra, we will ensure that the results of the work are communicated through reports and summaries and personal communications to other groups working on the programme.

Information on the project, in a format which can readily be understood by non-experts, will be summarised on the Stockholm Environment Institute website on completion of the project to the satisfaction of Defra. This will be linked to other relevant websites as agreed with Defra.

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