Using the Ecosystems Services Approach to Value Air Quality

(Defra Project NE0117)

Report to Department for Environment Food and Rural Affairs (Defra)

2012

Jones, M.L.M.[1], Provins, A.[2], Harper-Simmonds, L.[2], Holland, M.[3], Mills, G.[1], Hayes, F.[1], Emmett, B.A.[1], Hall, J.[1], Sheppard, L.J.[4], Smith, R.[4], Sutton, M.[4], Hicks, K.[5], Ashmore, M.[5], Haines-Young, R.[6]

This document has been prepared by:

[1] Centre for Ecology and Hydrology (CEH)
Environment Centre Wales
Deiniol Road
Bangor
LL57 2UW

In association with:

[2] Economics for the Environment Consultancy Ltd (eftec)
73-75 Mortimer Street
London
W1W 7SQ

and

[3] EMRC
2 New Buildings
Whitchurch Hill
Reading
RG8 7PW

with contributions from

[4] Centre for Ecology and Hydrology (CEH), Bush Estate, Edinburgh.
[5] Stockholm Environment Institute at York, Grimston House, University of York, Heslington, York, YO10 5DD.
[6] Fabis Consulting, 13 Chestnut Lane, Barton-in-Fabis, Nottingham, NG11 0AE.

Please cite this document as:
Jones, M.L.M., Provins, A., Harper-Simmonds, L., Holland, M., Mills, G., Hayes, F., Emmett, B.A., Hall, J., Sheppard, L.J., Smith, R., Sutton, M., Hicks, K., Ashmore, M., Haines-Young, R. (2012). Using the Ecosystems Services Approach to value air quality. Full technical report to Defra, project NE0117.

(Formerly cited as: Jones, M.L.M., Provins, A., Harper-Simmonds, L., Holland, M., Mills, G., Hayes, F., Emmett, B.A., Hall, J., Sheppard, L.J., Smith, R., Sutton, M., Hicks, K., Ashmore, M., Haines-Young, R. (2012 Draft). Using the Ecosystems Services Approach to value air quality. Draft Full technical report to Defra, project NE0117.)

EXECUTIVE SUMMARY

Objectives and scope

Air pollution has considerable impact on the natural environment *via* processes such as eutrophication of terrestrial and aquatic ecosystems, acidification of soils and freshwaters, and direct toxicity effects of ground level ozone. These ecological impacts affect supporting ecosystem services, with consequent effects on final provisioning, regulating and cultural ecosystem services, and the goods and benefits derived from them. The conceptual framework of 'ecosystem services' is increasingly recognised as providing a basis for quantification and ultimately valuation of many aspects of the benefits we derive from the environment.

The aim of this study was to apply an 'ecosystem services approach' (ESA) to value the impacts of air pollution on the natural environment. The three main objectives were:

i). Estimate the economic value of environmental impacts arising from changes in emissions/concentrations of nitrogen oxides (NO_x), sulphur oxides (SO_x) ammonia (NH_3) and ozone (O_3);
ii). Provide an indicative methodology for valuing impacts on the natural environment from air pollution; and
iii). Assess and identify research gaps with respect to objectives (i) and (ii) and provide recommendations to address these.

The study focused on a selection of ecosystem services in order to test the ecosystem services approach. Therefore this is not a comprehensive valuation of the total economic impact of air pollution on the natural environment. Instead, it is an application of the ESA methodology and demonstrates the range of both positive and negative impacts that may be observed on ecosystem services.

The economic value of air pollution impacts on these services is estimated using value transfer from existing valuation studies. The analysis is presented in broad terms at national level, with emphasis placed on understanding the main links between emissions and subsequent impacts on ecosystem services. These quantified results should be interpreted as generalised and indicative at the UK scale, rather than representative of site-specific effects.

Approach – impact pathway approach and ecosystem services framework

The established methodology for assessing the impacts of air pollution is the 'impact pathway approach' which traces the chain of causal relationships from the source of air pollutant emissions to changes in atmospheric air quality and subsequent impacts on human health and ecosystems. In this study, we combined the impact pathway approach with an ecosystem services approach to strengthen understanding of how air pollution affects provision of ecosystem services across a range of UK habitats. The generalised impact pathway used was defined to include:

- Impacts of policy on pollutant emissions/concentrations;
- Associated changes in pollutant deposition;
- Air pollution impacts on ecosystem processes in multiple ecosystems;
- Impacts on the final ecosystem services and the goods and benefits delivered by those ecosystems; and

- Valuation of the effect on social well-being via market and non-market impacts.

A subset of six goods and benefits from the UK National Ecosystem Assessment list of final ecosystem services was selected for valuation, in consultation with the project steering group. These included two each from provisioning services (timber production; livestock production), regulating services (net greenhouse gas emissions: CO_2 sequestration, methane and nitrous oxide emissions; water quality) and cultural services (appreciation of biodiversity; recreational fishing). Impacts were evaluated across as many relevant habitats as possible for each service.

Air pollution trends, and description of scenarios

The air pollutants evaluated in this study show varying trends in emissions and deposition. Sulphur emissions and deposition have declined strongly since 1970 and are projected to further decline. Nitrogen dioxide emissions and, to a lesser extent, deposition have also declined since around 1990 and are projected to decline further. Ammonia emissions and deposition have declined a little since 1990, with only minor reductions in both emissions and deposition projected beyond 2010. Ozone trends show an increase in background concentrations, but declines in the severity of peak episodes. On average, UK ozone concentrations have risen since 1987, and are projected to rise more strongly to 2020.

Two scenarios were chosen as a basis for economic valuation in this study:

- ‘***Historic emissions scenario***’: based on observed emissions/concentrations for the period 1987 – 2005, using 1987 as baseline; i.e. what would be the difference in ecosystem service value if current levels of air quality had not been achieved?
- ‘***Projected emissions scenario***’ based on projected emissions/concentrations for the period 2005 – 2020, using 2005 as baseline and assuming linear trends in emissions/concentrations; i.e. looking forward, what is the expected impact on ecosystem service values if projected changes in air quality are not achieved, compared with current levels?

Nitrogen and sulphur emissions data were from UK emissions inventories, including projections for 2020 under the UEP30 emissions scenario; and deposition data were from CBED data for historical deposition and FRAME model outputs for 2020, spatially calibrated to CBED data, all summarised in RoTAP (2010). Ozone AOT40 data were calculated by CEH from monitoring network data. Ozone data from 2008 (a typical year) and 2006 (a high ozone year) were used as proxies for 2005 and 2020 ozone regimes respectively. Linear trends in nitrogen, sulphur and ozone were assumed where observed/modelled data were not available.

Dose response functions linked impacts of nitrogen and sulphur to the quantity of pollutant deposition. Ozone response functions were linked to the concentration-based measure AOT40, rather than the newer, biologically more relevant, flux-based PODy approach due to data availability. All calculations used UK average deposition/concentration data and were not spatially explicit.

For each scenario, the difference in pollutant deposition/concentrations and consequent impact were calculated for each year relative to the baseline of no change in deposition. Economic values, estimated via value transfer, were discounted and equivalent annual value calculated according to Green Book guidance (HM Treasury, 2003).

Uncertainty analysis of the impact pathways was based on Monte Carlo approaches. Results are presented as central estimates, with lower and upper 95% Confidence Interval bounds.

Results – estimated value of air pollution impacts

The valuation of air pollution trends on ecosystem services revealed both positive and negative impacts, differing by pollutant and ecosystem service. Impacts were summed for each pollutant for the services considered but were not summed across all pollutants for two reasons. Firstly, combined impacts may not be equivalent to the sum of their individual effects. Secondly, while nitrogen and sulphur pollution have been declining, average ozone concentrations have been rising, and these are valued separately.

Of the six ecosystem services selected for study, it was not possible to value impacts on all ecosystem services across all habitats, and many gaps remain (**Table E1**). Nitrogen impacts were the most comprehensively valued across the widest range of habitats. It was only possible to comprehensively value one service for sulphur and two services for ozone, across a more limited range of habitats. Impacts of reductions in nitrogen deposition under the projected emissions scenario are summarised in **Figure E1**.

Table E1: Habitats and services where valuation was possible for each pollutant. n.v. = not valued in this study.

	Provisioning Services		*Regulating Services*				*Cultural Services*	
			Net GHG emissions					
.	Timber production	Livestock	CO_2	N_2O	CH_4	Clean water	Recreational fishing	Appreciation of biodiversity
Nitrogen	Woodland	Improved grassland: Partially valued	Woodland, Heathlands	All semi-natural habitats	n.v.	n.v.	Upland rivers: Partially valued	Woodland, Heathland, Grasslands and Bogs.
Sulphur	n.v.	n.v.	n.v.	n.v.	Bogs	n.v.	n.v.	n.v.
Ozone	Woodland	n.v.	Woodland, Grasslands	n.v.	n.v.	n.v.	n.v.	n.v.

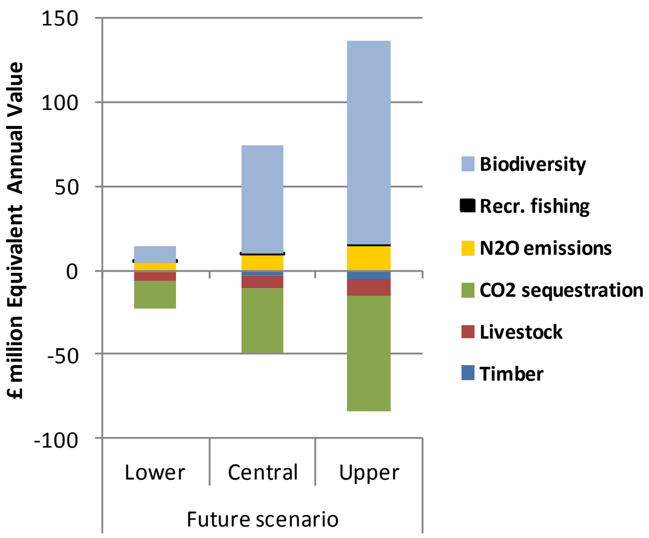


Figure E1: Summary of costs (negative) and benefits (positive) from reductions in nitrogen emissions (£million Equivalent Annual Value - EAV), by ecosystem service, for the projected emissions scenario (2005 to 2020). Lower, Central and Upper represent central estimate and the lower and upper 95% confidence intervals from the uncertainty analysis.

Declines in nitrogen deposition result in a loss of ecosystem service value for timber production, livestock production, and carbon sequestration, but a gain in ecosystem service value for emissions of the greenhouse gas nitrous oxide, recreational fishing and biodiversity. The net EAV is £65.8m (£5.1m to £123.2m, 95% CI) and £24.6m (-£9.2m to £52.7m, 95% CI) per year for the historical and projected emissions scenarios respectively. The large range around the best estimates indicates the considerable uncertainty present in this analysis, particularly for biodiversity (the valuation of which is subject to very high levels of uncertainty), and it may be possible to narrow the upper and lower range bounds with further work.

Declines in sulphur deposition result in a loss of ecosystem service value for methane emissions. The net EAV is £-1.1m (-£0.4m to -£2.1m, 95% CI) and £-0.9m (-£0.3m to -£1.8m, 95% CI) in the historical and projected emission scenarios. However, we were only able to value one service in this study and the costs are likely to be outweighed by considerable benefits to other ecosystem services such as appreciation of biodiversity and recreational fishing.

Increases in average ozone concentrations result in a loss of ecosystem service value for carbon sequestration and timber production. Net EAV is -£2.6m (-£1.7m to -£3.6m, 95% CI) per year and -£11.3m (-£7.2m to -£14.7m, 95% CI) in the historical and projected emission scenarios. It was only possible to value two services in this study, using AOT40 rather than the more biologically relevant flux-based approach for quantifying impacts. Further losses could potentially arise from damage to livestock production and to biodiversity.

The impact varies across category of ecosystem service. For example, declines in nitrogen have a negative impact on provisioning services since it is a nutrient which stimulates plant growth. Conversely, declines in nitrogen have a positive impact on cultural services such as biodiversity and recreational fishing due to alleviation of habitat damage. Outcomes for the regulating service greenhouse (GHG) emissions were mixed, with both gains (nitrous oxide emissions) and losses (carbon sequestration and methane emissions) evident.

Estimation of damage costs

Based on the results reported above, unit damage benefits/costs were separately calculated per tonne reduction in emissions of NO_x, NH_3, and per unit increase in ozone 6-month AOT40 (ppmh). Net damage costs for sulphur were not calculated due to insufficient valuation data. Overall results should be interpreted as indicative of damage costs and are presented for comparative purposes only. <u>These damage costs and subsequent work under other contracts have been collated in report AQ0827, together with an assessment of their robustness (Jones et al. 2014). That report should be the main reference source for information on damage costs</u>. It is important to recognise that pollutant deposition is a function of both pollutant import, changes in UK pollutant emissions and pollutant export. In this study it was not possible to separately calculate the impact of deposition changes resulting solely from UK emissions, and damage costs are calculated per tonne of pollutant emitted in the UK.

For NO_x the net damage cost estimates for impacts on ecosystem services reported here are a benefit of £77 (-£48 to £196, 95% CI) per tonne <u>reduction</u> in NO_x emissions in the historical emissions scenario, and £22 (-£97 to £129, 95% CI) per tonne <u>reduction</u> in the projected emissions scenario. For NH_3 the net damage cost estimates are a benefit of £692 (-£156 to £1,526, 95% CI) per tonne <u>reduction</u> in NH_3 emissions in the historical emissions scenario, rising to £1,246 (-£2,227 to £4,559, 95% CI) per tonne <u>reduction</u> in the projected emissions scenario. Net damage costs for ozone are a loss of -£9.1 million (-£5.8m to -£12.4m, 95% CI) per unit <u>increase</u> in 6-month AOT40 (ppm hours) in the historical scenario and -£11.4 million (-£7.3m to -£15.5m, 95% CI) per unit <u>increase</u> in the projected emissions scenario.

These are net damage costs, incorporating both costs and benefits. Damage costs for some ecosystem services, e.g. biodiversity (which is subject to very high uncertainty), are higher than the net totals.

The estimated unit values for ozone for two ecosystem services in the marginal cost analysis here are roughly one sixth of those calculated for seven key crops in the UK in 2006 and 2008 (Mills et al. 2011b): -£60m to -£66m per unit increase in AOT40 (ppm h) (accumulated over 3 months, centred on the main growing season for each crop, and based on total ozone impact compared to a zero AOT40 reference). The unit values for nitrogen compounds are lower than IGCB damage costs for human health, for NO_x (£955/t) and NH_3 (£1,972/t), 2010 prices.

It is reasonable to expect that the indicative damage costs may be 'under-estimates' due to the partial coverage of ecosystem services. As additional ecosystem services are valued, particularly regulating and cultural services which were not valued in this study but where air pollution impacts are likely to be negative, the damage costs may be expected to increase. In addition, based on current trends, as UK nitrogen emissions decline a smaller proportion will be exported – particularly of ammonia - and any declines in nitrogen emissions will be more immediately manifest in declining deposition; i.e. the change in impact per unit pollutant emitted is likely to be higher. However, it should be re-iterated that the indicative damage costs for biodiversity are highly uncertain and may alter pending assessment of the assumptions linking damage to valuation.

As is evident from the reporting here, all results are subject to a number of assumptions and caveats. Principally, these arise from: (i) the specification of the emissions scenarios; (ii) the available scientific evidence and its application; and (iii) the available economic value evidence and its application; and (iv) the lack of spatial analysis of impacts within this study. The uncertainty analysis captures some of these issues, but does not always capture structural uncertainty where the evidence base is limited. Detailed discussion of the limitations of the analysis is presented in the Main Report and Annex 3 (Jones et al. 2012 Annex 3).

Conclusions – evidence gaps and recommendations

The study successfully demonstrates that the ecosystem services approach aligned with the impact pathway approach can be used to evaluate air pollution impacts on ecosystem services. It illustrates the practical application of a methodology for valuing the ecosystem service impacts of air pollution, and in doing so, a number of evidence gaps and limitations have been identified:

- Gaps in the scientific evidence limit the practical specification and quantification of some impact pathways.
- The broad brush approach has quantified effects based on total deposition or mean concentration across the UK and total ecosystem service value. It does not consider the spatial context of pollutant impact or valuation across the UK.
- Air pollution effects are dependent on multiple ecological processes and the complex interactions between them. The resultant impacts on the 'final' ecosystem services and goods utilised by human populations are typically several steps removed from these initial effects, increasing the potential for gaps in knowledge.

- In general, of the pollutants considered, understanding and evidence related to eutrophication from nitrogen deposition has permitted the widest valuation of impacts across various semi-natural habitats. Quantification of impacts with respect to acidification and direct toxicity is more limited.

- Some potentially significant impacts have been excluded from the analysis due to an explicit lack of understanding of complex ecological systems. These include uncertainty concerning the ecological interactions governing methane emissions (in relation to GHG emissions) and relationships between deposition of atmospheric pollution and changes in the ecological status of the water environment.

- Limitations also arise through extrapolation of dose-response relationships for particular contexts (e.g. a species) to more generalised assessments at a national habitat scale. In some cases there is a risk of applying site and species specific dose-response functions that are not representative of wider scale effects.

- The analysis with respect to provisioning services (timber and livestock) highlights the need for a sufficient understanding of land use management responses to air pollution impacts. These are site-specific and based on the management objectives of land managers. Generalising these responses across a broad brush assessment requires use of simplifying assumptions and consequently reported results need to be carefully interpreted.

- In a number of cases, especially appreciation of biodiversity and recreational fishing, the analysis employs several assumptions in linking scientific evidence to available economic valuation, which result in very high levels of uncertainty in reported value estimates. These should be viewed as key areas for refinement of the evidence base that supports this study.

- To summarise, this study successfully demonstrates that ecosystem services can be used to evaluate impacts of air pollution. A number of assumptions are employed; however key among these are the value-transfer evidence linking habitat damage to willingness to pay for biodiversity.

Recommendations for addressing these research gaps are set out in the Main Report. In summary, they are:

- **Future research**: several specific gaps are identified in relation to eutrophication, acidification and toxicity impacts. As a first step it is necessary to prioritise the research gaps in order that research focuses on the elements likely to be most informative, either in estimation of total damage, or method development. It may be efficient to link these in to broader research programmes – potentially at the European level.

- **Specific research gaps:** A number of areas have been identified where significant progress is readily achievable. These include: Assessment of the uncertainty in dispersion modelling and exposure to pollutant concentrations; spatial quantification of pollutant deposition attributable to UK emissions; modelling to allow reliable upscaling of catchment acidification and effects on fish populations to the national scale; Improved quantification and valuation of nitrogen, ozone and acidity effects on biodiversity. Spatially explicit calculation of air pollution impact and valuation may be possible for some services.

- **Better design of research and outputs**: a challenge also exists in providing better and more appropriate scientific input for economic valuation exercises. A general recommendation is that scientific assessments developed to support policy and project analyses should not be seen as isolated tasks but part of the evidence that is developed in an ecosystem services framework. This perspective will also help commissioning of research, through better understanding of where the key gaps are.

- **Value transfer evidence:** the scope for undertaking robust valuations of ecosystem service values would be improved by developing further the available evidence base. This includes both market and non-market values across the range of ecosystems services. Where opportunities exist with respect to new studies, emphasis should be placed on developing 'transferable' value transfer tools. For provisioning services (e.g. timber, livestock) there is particular potential to explore the use of production function models that control for the influence of various human, physical and environmental factor inputs on the production of final goods. We also recommend further research on issues pertaining to valuation of biodiversity.

- **Review of value transfer evidence for biodiversity**: The greatest uncertainty in this study is around the value transfer evidence for appreciation of biodiversity, in particular, the alignment of the critical load exceedance scenarios and the BAP scenarios of Christie et al. (2010). Note, an improved approach is presented in report AQ0827 (Jones et al. 2014).

- **Increased use of multi-disciplinary expertise:** A broad multi-disciplinary team was essential in this study. Economic valuation represents the 'final step' in the qualitative – quantitative – monetary assessment process that underlies the ecosystem services and impact pathway approaches. For a consistent evidence base to be developed, the scope of each individual component should be viewed in this wider context. This requires dialogue between policy makers, scientists and economists to establish the requirements for policy and project analyses, so that current evidence needs are fulfilled and future gaps in evidence can be addressed.

CONTENTS

SEPARATE SUPPORTING DOCUMENTS

Annex 1. Review of available evidence and outline methodology (Jones et al. 2012 Annex 1)

Annex 2. Prioritisation of ecosystem services and representative ecosystems for valuation (Jones et al. 2012 Annex 2)

Annex 3. Treatment of Uncertainty and Value Transfer Analysis (Jones et al. 2012 Annex 3)

1. INTRODUCTION

1.1 Background

Air pollution impacts on human health are well documented and a number of studies have sought to estimate the economic value of associated mortality and morbidity impacts on affected populations (ExternE, 2005; CAFE, 2005; Defra, 2006). Air pollution impacts on the natural environment - via processes such as eutrophication of terrestrial and aquatic ecosystems, acidification of soils and freshwaters, and direct toxicity effects of ground level ozone - have been studied for some time. However, outside of agricultural crops and timber production, there have been few attempts (e.g. Hornung et al, 1995; Karlsson et al, 2004) to place an economic value on these impacts. In particular, to follow the impact pathways from emissions and atmospheric concentrations of air pollutants through to physical effects on ecosystems and the consequent value of those effects. Alternative approaches for valuing impacts on the environment have been trialled, but failed to gain acceptance[1]. Ecosystem Services are increasingly recognised as a conceptual framework which allows quantification and ultimately valuation of many aspects of the benefits we derive from the environment (TEEB[2]; UK NEA[3]).

In the last few decades, policy initiatives have sought to limit emissions of air pollutants and protect terrestrial and aquatic ecosystems from adverse impacts. The principal national and international policies relevant to UK air quality are:

- The EU National Emission Ceilings (NEC) Directive[4], currently under revision, sets upper limits for each Member State for the total emissions for the pollutants responsible for acidification, eutrophication and ground-level ozone pollution (sulphur dioxide, nitrogen oxides, volatile organic compounds and ammonia);
- The UNECE Convention on Long Range Transboundary Air Pollution[5] sets emission ceilings for the same pollutants to the NEC Directive via a multi-lateral agreement between EU Member States and other European countries, the United States and Canada (the Gothenburg Protocol, also currently under revision); and
- The Air Quality Strategy for England, Scotland, Wales and Northern Ireland (Defra, 2007) sets out objectives and measures to improve air quality in part based on critical levels and critical loads of pollutants. Exceedance of critical levels (pollutant atmospheric concentration) and of critical loads (quantitative measure of pollutant) is a measure of potential adverse effects on ecosystems.

It is recognised that declines in many air pollutants have occurred as a direct result of these policy initiatives. However, not all pollutants have declined, for example, mean ozone concentrations have risen over the last 20 years, due largely to long-range transport of precursor chemicals from other countries. Further, it is recognised that there are both costs and benefits associated with declines in air pollutants, and these have not been systematically evaluated to date.

[1] For example, a repair cost approach was investigated as part of the EC-funded NEEDS project. However, such an approach makes considerable assumptions, not least that repair is possible and meaningful, and that society is willing to pay for it.
[2] See http://www.teebweb.org/
[3] See: http://uknea.unep-wcmc.org/
[4] Directive 2001/81/EC: http://ec.europa.eu/environment/air/pollutants/ceilings.htm
[5] See: http://www.unece.org/env/lrtap/

1.2 Objective and scope

The aim of this study is to apply an 'ecosystem services approach' to valuing the impacts of air pollution on the natural environment. The Terms of Reference[6] for the study set out three main objectives:

i). Estimate the economic value of environmental impacts arising from changes in emissions of nitrogen oxides (NO_x), sulphur oxides (SO_x) ammonia (NH_3) and ozone (O_3);
ii). Provide an indicative methodology for valuing impacts on the natural environment from air pollution; and
iii). Assess and identify methodological and evidence gaps with respect to objectives (i) and (ii) and provide recommendations to address these.

The scope of the work is a secondary analysis utilising the currently available evidence base. The focus is a pragmatic application of the principles that underlie an ecosystem services approach and the valuation of environmental impacts. Necessarily the analysis is presented in broad terms at the national level, with emphasis placed on establishing an understanding of the main links between emissions and subsequent impacts. Results should be interpreted as generalised and indicative, rather than representative of site-specific effects.

The nature of the work means the study provides an opportunity to develop and trial an ecosystem services approach. While the analysis quantifies and values the impacts of air pollution, it is also intended to make progress in addressing many of the practical challenges faced when integrating scientific understanding and evidence of the effect of pollutants on ecological systems with information on economic values of resulting impacts in a multi-disciplinary setting.

The analysis is not intended to provide a comprehensive overview of the effects of pollutants on ecosystems or on ecosystem services. The purpose of the work is instead to assess whether there is potential for making progress on an area of work that has been frequently discussed in relation to policy in the UK and elsewhere in Europe, to provide an outline illustration of methods and illustrative estimates of the magnitude of impacts.

1.3 Approach

The analysis and results presented in this report are the culmination of a number of discrete tasks that have been undertaken in the course of the study. The tasks provide the building blocks for compiling the available scientific and economic value evidence on the impacts of air pollutants and the practical application of this evidence via an ecosystem services approach:

1. Review air pollution effects on ecosystem services – identify and summarise available information;
2. Select representative ecosystems of concern – based on assessment of air pollution critical loads and exceedances, and conceptual understanding of major air pollutant impacts;
3. Select a set of ecosystem services, likely to be affected by air pollution via a prioritisation exercise;
4. Develop an approach for valuing impacts of air pollution on selected ecosystem services across those ecosystems; and
5. Estimate the value of air pollution impacts.

[6] Defra Invitation to Tender: Provision of Research Project to Use the Ecosystem Services Approach to Value Air Quality, NEE 1001.

Reporting in this document focuses on tasks 4 and 5 set out above. Supporting annexes detail outcomes from tasks 1-4.

1.4 Structure of report

The remainder of the report is structured as follows:

- Section 2: provides a conceptual overview of the valuation of ecosystem services and air pollution impacts, detailing the main principles that inform the approach utilised in the study;
- Section 3: sets out the practical details of the approach used to value the impacts of air pollution, including details of air pollution trends and the scenarios applied;
- Section 4: reports results from the analysis for the selected ecosystem services across the selected ecosystems, including sensitivity analysis of the valuations;
- Section 5: discusses the practicalities of applying the ecosystem services approach, highlights key knowledge gaps, and presents the study's conclusions including recommendations for addressing identified gaps in the evidence base.

Supporting annexes include:

- Annex 1: review of air pollution effects on ecosystem services, covering both the scientific and economic valuation evidence (Jones et al. 2012 Annex 1);
- Annex 2: selection of ecosystem services and representative ecosystems (Jones et al. 2012 Annex 2); and
- Annex 3: reporting of value transfer analysis and results, including methodology for uncertainty analysis (Jones et al. 2012 Annex 3).

A subsequent Defra report AQ0827 collates damage costs from this and other studies (Jones et al. 2014).

2. CONCEPTUAL OVERVIEW

2.1 Valuing ecosystem services

2.1.1 Background

In recent years the ecosystem services approach has been the focus of concerted research effort and policy analyses which have highlighted and attempted to measure the contribution of ecosystems and the biological diversity contained within them to individual and social wellbeing. In this sense, the term 'ecosystem services approach' has come to describe a basis for analysing how individuals and human systems are dependent upon the condition of the natural environment.

In practice however, there is no single ecosystem services approach or framework, and different interpretations of the approach are taken. While numerous research initiatives have been undertaken it is widely recognised that the key contribution in developing a high profile systematic account of ecosystem services was provided by the UN Millennium Ecosystem Assessment (MEA, 2003; 2005). Subsequent studies have sought to improve understanding, refine concepts and develop practical applications of ecosystem service approaches. Recent major initiatives include 'The Economics of Ecosystems and Biodiversity' (TEEB) and the 'UK National Ecosystem Assessment' (UK NEA). A substantial academic literature has also developed particularly with respect to the role of economic analysis within the ecosystem services approach[7]. The Ecosystem Services Approach (ESA) as defined by Defra (2007) identifies three key steps:

- Identifying which ecosystem services will be affected by a policy decision.
- Prioritising those ecosystem services, including consideration of environmental limits, designated sites and species and other regulatory factors.
- Valuing the benefits obtained from those ecosystem services, and which will be affected by a policy decision.

The ecosystem services approach has played a useful role in highlighting the fact that valuation needs to be underpinned by scientific understanding and assessments of the provision of ecosystem services. Much emphasis is placed on the multi-disciplinary input required from environmental science and economics disciplines. In turn, this has contributed to improved understanding of the role of valuation across both disciplines and in the wider policy arena.

2.1.2 Economic valuation of ecosystem services

From an economic perspective, ecosystem services represent flows of economic value that are generated by stocks of ecological assets; i.e. 'natural capital' (Bateman et al., 2011). For example timber represents a flow of benefits that can be realised from forests. Numerous ecological functions and processes (e.g. nutrient cycling) influence the types of ecosystem services that are derived from ecosystems. Various ecosystem services contribute to the production of market and non-market goods, the consumption of which generates human 'wellbeing'. The UK NEA makes a particular distinction between a 'good' – items which generate wellbeing – and a 'benefit'. In particular the term benefit is applied to changes in the wellbeing that are generated by the consumption of goods, the economic value of which can be context-specific and dependent on factors such as spatial location and timing. Benefits derived from ecosystem services are also

[7] For example: Balmford et al., 2002; de Groot et al., 2002; Heal et al., 2005; Fisher et al., 2008; Mäler et al., 2008; Turner et al., 2010.

distinguished by the familiar typology of 'total economic value' (TEV) which recognises that they can be attributed to use and/or non-use values.

In practical terms economic valuation represents the final step in the qualitative – quantitative – monetary assessment process of project and policy analyses (Defra, 2010). In order to estimate the value of change in provision of a market or non-market good, the following information is needed (eftec, 2010):

1) ***An estimate of the change in the provision of the good under consideration***: This can be presented in qualitative and/or quantitative terms. While this is the first step in the process of economic valuation, it requires considerable scientific input and detailed understanding of the environmental processes which underpin ecosystem function. It also requires scientific, social and economic understanding of how changes in ecological function translate into impacts on ecosystem services, a step that is only recently being addressed in the scientific literature. Robust conceptual links, supported by published evidence, enable the impact pathway to be followed through from physical impacts on ecosystem services (e.g. eutrophication arising from nitrogen deposition) to changes in the provision of goods consumed by user and/or non-user populations, in terms of changes in their quantity, quality, timing or spatial availability.

2) ***A reliable estimate of the economic value:*** The basic principles of economic valuation assert that economic value is measured by the resource individuals are willing to trade-off to either secure or forego the change in provision of a good. Ordinarily the 'resource' is defined in terms of money and economic value can be estimated via the metrics of 'willingness to pay' (WTP) or 'willingness to accept' (WTA) (Bateman, 2007). Economic valuation methods estimate WTP or WTA using different types of data depending on whether the good is traded in actual markets or not; e.g. market prices, revealed preference methods and stated preference methods (Hanley and Barbier, 2009). For market goods, the direct use value associated with their provision can be reflected by market price information where this represents opportunity cost (i.e. net of distortions such as taxes and subsidies). Valuation of non-market goods relies on revealed and stated preference methods in terms of primary studies, and more commonly in project and policy analyses, value transfer approaches (see Section 2.1.3). Different methods are able to capture to differing extents the components of total economic value; i.e. direct use values, indirect use values, and non-use values (see for example: Defra, 2007; eftec, 2006).

3) ***Knowledge of how the economic value (2) changes due to the change in provision of the good (1):*** In many instances, it is not sufficient to simply assume that there is a constant relationship between economic value and changes in the provision of market and/or non-market goods. For example the value of improvements in environmental quality can be subject to diminishing marginal utility, implying that benefits from initial improvements are valued greater than subsequent improvements. This highlights the context-specific nature of economic values and particularly how they are dependent on the baseline provision of the good and the scale of the change in provision to be valued.

4) ***Knowledge of which factors influence the economic value:*** In addition to the scale of the change in provision of a good, the context-specific nature of economic values is also dependent upon the circumstances of the population which benefits from its provision. For example the abundance and quality of substitute goods is a fundamental determinant of demand for both market and non-market goods. Willingness to pay of individuals – the most commonly applied metric for valuing non-market goods – is also dependent upon the socio-economic characteristics (e.g. income) and patterns of use of goods by user populations (or, for example, familiarity with the good for non-user populations). Significantly these factors are 'spatially sensitive' - i.e. they vary over populations and their spatial distribution - implying that

economic values and consequently benefits derived from the provision of ecosystem services are also spatially sensitive. This gives rise to empirically observed relationships such as 'distance decay', which refers to a decline in use value, and the user proportion within the population, as distance from a resource (e.g. a recreation site) increases (Bateman et al., 2006).

Overall, the integration of economic valuation within the framework of the ecosystem services approach provides a basis for establishing and assessing the range of impacts associated with project and policy initiatives. In particular it provides a transparent process for establishing the value of changes in provision of goods that are generated in-part or wholly by ecosystem services. Much emphasis has been placed on the care needed to avoid the risks of double counting in valuation by initiatives such as the UK NEA and TEEB[8]. This is reliant on the necessary scientific understanding and assessments that contribute to multi-disciplinary analysis, requiring them to work towards identifying the 'final' ecosystem services that provide the market and non-market goods which confer economic value to affected populations.

2.1.3 Use of value transfer

Value transfer is an approach to economic valuation that utilises existing and readily available economic valuation evidence. Specifically, economic value evidence estimated in one context is applied ('transferred') to a similar context for which valuation is required. In practice there are several different ways in which value transfer can be applied. Guidelines provided by Defra (2010) highlight the degree of complexity, data requirements and expected reliability of the results associated with different approaches; i.e. unit value transfer, adjusted unit value transfer, and function transfer.

The scope of this study is well suited to the use of value transfer. Principally the key aims of the work are to utilise existing scientific and economic evidence to demonstrate how impacts from air pollution on the natural environment can be assessed in an ecosystem services framework. This aligns well with value transfer principles that require explicit account to be made of the overall policy context within which value transfer will be applied. Limitations of value transfer include:

- There can be a scarcity of suitable studies from which to source valuation evidence;
- There are likely to be 'transfer errors' when evidence from an existing study is applied in a new context and the level of error may be unknown; and
- Selection and adjustment of the value evidence involves a degree of expert judgement and can entail assumptions that are not consistent across assessments undertaken by different analysts.

In the case of this study the objective is to provide indicative estimates of the value of air pollution impacts on the provision of ecosystem services. Emphasis is on addressing the methodological steps and challenges in linking scientific and economic evidence within a coherent framework, and from this, establishing key evidence gaps and recommendations to address these. As such the context for applying value transfer is that of 'gains in knowledge' and demonstrating the likely significance of air pollution impacts on selected ecosystem services.

[8] It is perhaps notable that the potential for double counting of impacts often appears to be taken more seriously in economic analysis than its converse – the omission of one or more effects.

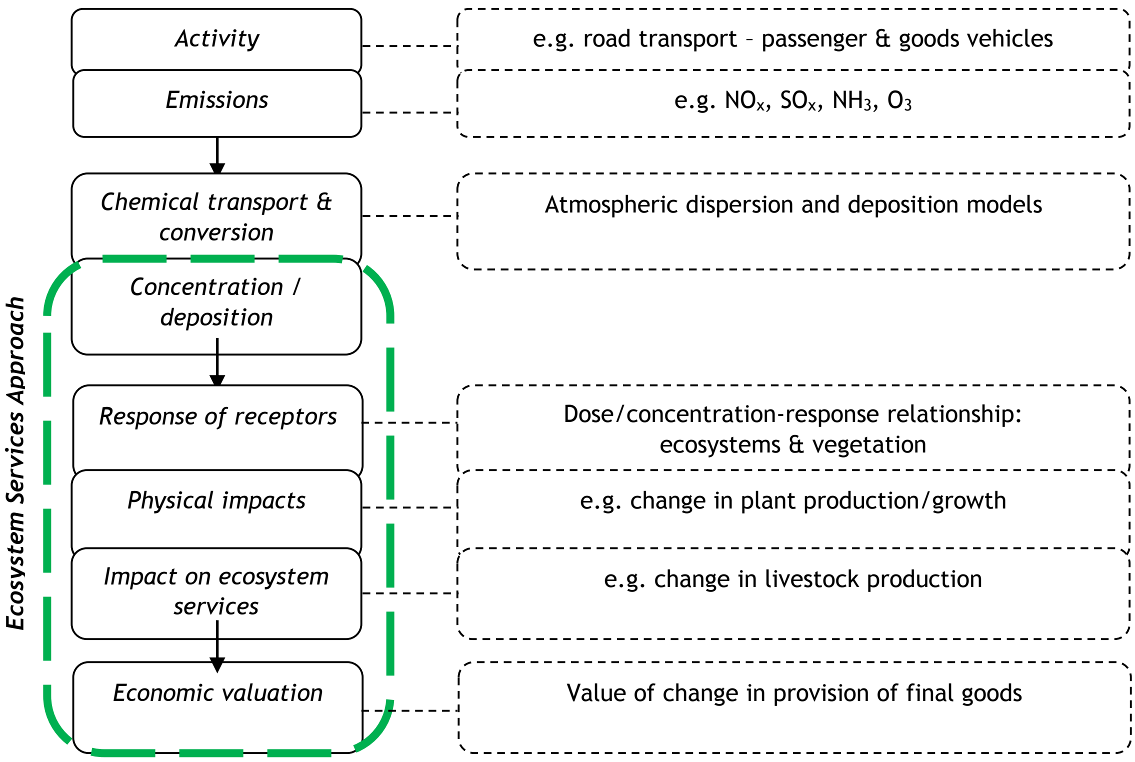
2.2 Assessing the impacts of air pollution on ecosystem service provision

2.2.1 Impact pathway approach

The established methodology for assessing the impacts of air pollution is the 'impact pathway approach'. Initially developed through the ExternE project[9], the impact pathway approach has subsequently supported wider policy initiatives including the Defra (2006) Air Quality Strategy and the Clean Air for Europe (CAFE, 2005) programme. The main focus of these analyses has been the human health impacts of air pollution, although assessments have also included impacts on agricultural crop production (e.g. Vlachokostas et al., 2010; Holland et al., 2005).

The impact pathway approach is a 'bottom-up' methodology that traces the chain of causal relationships from the source of air pollutant emissions to changes in atmospheric air quality and subsequent impacts to receptors such as human populations, ecosystems, and buildings and materials, in order to establish the physical impacts (**Figure 2.1**). The value of these impacts – either in terms of costs from deterioration in air quality or benefits from improvements - can then be expressed in monetary terms through the application of economic valuation methods.

Figure 2.1: Relationships between impact pathway approach (whole chain) and ecosystem services approach (circled by green dashed line).



In 'ideal' applications the bottom-up approach implies that assessments measure marginal costs associated with specific changes in the emissions of air pollutants. Relating back to Section 2.1.2, this is significant since the marginal costs of pollution may change with the overall concentration of pollutant in the atmosphere; that is, the concentration response function may not be linear. Where response functions are not linear, the average cost method based on estimates of damage costs per tonne of emission may not accurately measure the possible costs or benefits from a policy

[9] The ExternE (Externalities of Energy) project is a European Commission funded research body. See: http://www.externe.info/

that increases or decreases the level of air pollution, respectively, if these ranges fall outside the ranges examined in the original study. In addition, the unit cost may well be different if pollutant levels are decreasing than when they are increasing due to hysteresis effects, and time-lags in recovery.

As illustrated in Figure 2.1 the impact pathway approach largely encompasses the ecosystem services approach and consists of:

- ***Emissions***: in a full assessment, the first step is to describe activity levels and technological standards over the timeframe of interest. These determine emissions from activities of interest, such as transportation or the energy generation sector. In general, the objective is to assess the difference in emissions arising under some proposed policy scenario (the 'with' case) and a continuation of the current situation (the 'without' case) which is also referred to as the 'baseline'. This can require detailed modelling; for example, in the case of road transport, aspects to consider include how the policy initiative changes behaviour (e.g. number of journeys, length and duration) and more dynamic effects such as how the emissions profile of the vehicle fleet changes over time. In addition, different types of emission source can require different analyses; e.g. a point source is a single source of emissions, such as a power plant; a line source is a one-dimensional source of emissions, such as traffic along a motorway; an area source is a two-dimensional source of emissions, such as a landfill, a road network, or town; and a volume source is a three-dimensional source of emissions, such as an oil refinery which emits pollutants at different heights. Spatial context is also important. Different constraints are associated with modelling changes in emissions from a few point sources than when modelling changes in diffuse pollution.

- ***Dispersion and deposition models***: in the second step of the pathway, atmospheric dispersion modelling predicts the concentration levels of a pollutant and subsequent deposition over a spatial area and how this changes with changes in emissions. In general, while the concentration of a pollutant resulting from an activity decreases as distance from the source of emissions increases, the total area affected increases, along with the number of potential receptors. Different pollutants have different residence times in the atmosphere, and also undergo chemical transformations, including interactions with other pollutants. A number of models exist which can be applied within a local or regional framework to estimate changes in the concentration of pollutants given various scenarios. At the UK scale, outputs from dispersion models are often calibrated to observed concentrations/deposition from pollutant monitoring networks. One example is the UK Concentration-Based Estimated Deposition (CBED) data for nitrogen and sulphur deposition, produced by CEH.

- ***Impacts***: information on the dispersion and deposition of pollutants can be mapped against databases of 'stock at risk' (forests, areas growing different crops, acid-sensitive waters, etc.), permitting 'at risk' receptors to be identified. This step then continues to quantify the physical impacts on receptors that occur due to the change in air quality (pollution). As noted much work to date has focused on human populations (Friedrich et al., 2001; CAFE, 2005; Defra, 2006). The concentrations and deposition of pollutants in different areas, as calculated by air quality models, can be translated into impacts through response functions, which estimate the relationship between the quantity of a pollutant and the scale of physical impacts. These can take a number of forms, being linear, non-linear or subject to threshold effects, and a function is required for the assessment of each impact on each receptor. Impacts may be viewed first within a group of receptors, such as human health, ecosystems and vegetation, and materials, and then at a more detailed level within each receptor; for example, within ecosystems, the environmental effects of a suite of pollutants will each have a variety of impacts on different ecosystem services, and these impacts will further differ by habitat type, requiring separate exposure response functions for each impact-service-habitat combination. To ensure they are as accurate as possible, these functions are derived from meta-analyses of relevant studies, where available, and are constantly being updated.

- ***Valuation of impacts*:** in the final step of the pathway the quantified physical impacts are valued in monetary terms using value transfer or other economic valuation methods.

Combining the impact pathway approach with an ecosystem services approach strengthens the understanding of how air pollutants impact on ecosystems. Figure 2.1 highlights that the ecosystem services approach fits around the process of establishing concentration and dose response functions, physical impacts and translating these into economic valuations, with the added step of translating physical/ecological impacts on the environment into specific effects on ecosystem services.

Annex 1 (Jones et al. 2012 Annex 1) reviews the available evidence in relation to the range of impacts that nitrogen, sulphur and ozone have on ecosystems via eutrophication, acidification and direct toxicity. It highlights that air pollutants act primarily on the processes which underlie the functioning of ecological systems, which are mostly closely aligned with the concept of 'supporting services'. Subsequent impacts on the final goods and beneficiary populations are typically several steps removed from these effects. They are ordinarily dependent on multiple processes and interactions that build from direct toxicity and cell damage, to indirect effects mediated by changes in individual organisms and their ecological interactions, and in the rate and nature of biological and chemical processes.

Establishing the quantitative relationships between changes in supporting services and the goods and benefits arising from 'final ecosystem services' is a key challenge that this study has had to address.

3. VALUING AIR QUALITY IMPACTS ON ECOSYSTEM SERVICES

3.1 Outline of value transfer approach

3.1.1 Value transfer steps

The approach taken to value air pollution impacts on the provision of ecosystem services combines the three main concepts outlined in Section 2: the ecosystem services approach, the impact pathway approach; and value transfer. In practice these are complementary concepts. The ecosystem services and impact pathway approaches are frameworks for establishing the scale and significance of impacts on the natural environment. Both rely on the same scientific evidence and assessments and work through to establishing how physical impacts affect the wellbeing of beneficiary populations via the provision of market and non-market goods. The end-points then require the use of either primary valuation methods or value transfer.

The basis for valuing impacts on ecosystem service in this study is the Defra (2010) value transfer guidelines. These set out eight steps establishing the nature of physical impacts on the environment and identifying and applying appropriate economic valuation evidence:

1. Establish the policy good decision context
2. Define the policy good and affected population

3. Define and quantify the change in provision of the policy good
4. Identify and select economic valuation evidence
5. Transfer evidence and estimate the value of the policy good
6. Aggregation
7. Conduct sensitivity analysis
8. Reporting

In the terminology of value transfer, the 'policy good' refers to the impact that is to be valued; i.e. the change in provision of final goods resulting from the impact of air pollution of ecosystem services. The following discusses key points for the methodological approach for the study under Steps 1 - 7. Step 7 is extended beyond 'sensitivity analysis' to include a more complete assessment of uncertainties, in keeping with that used to inform development of the Air Quality Strategy.

3.1.1 Policy good decision context

As highlighted in Section 1.3 a key aim of the study is to test the use of an ecosystem services approach in relation to assessing impacts of air pollution on the natural environment. The Terms of Reference for the work require that the analysis quantify the accumulated damage to ecosystem services from air pollution, and provide a snapshot of damage in a single year. This requirement is addressed via the specification of two hypothetical scenarios that measure historic and projected emissions of nitrous oxides (NO_x), ammonia (NH_3), sulphur oxides (SO_x), and ozone (O_3) against a common reference point. This permits the estimation of a notional change in the flow of ecosystem service on an annual basis relative to the reference point – see Section 3.1.3. The analysis represents a broad-brush approach at the national level rather than detailed site-specific assessments for particular pollutants or ecosystem services. Comparing the relative magnitudes of changes in economic values associated with final goods permits a judgement as to the scale and significance of air pollution on the provision of different ecosystem services that are included in the analysis.

3.1.2 Policy good definition and affected population

The analysis focuses on seven market and non-market goods[10] provided by final ecosystem services which were selected from the UK NEA (2011) list of final services (Table 3.1).

[10] In most cases the provision of these goods and the benefits derived by human populations are reliant on natural capital, physical capital and human capital inputs (e.g. harvesting of timber requires physical capital inputs). In line with Bateman et al. (2011) and principles set out in the UK NEA the analysis attempts to 'isolate' and value the ecosystem service contribution to the wellbeing derived from the consumption of these goods. However the ability to do this is constrained by the availability of relevant data and information, which is discussed on a case-by-case basis in Annex 3.

Table 3.1. Final Ecosystem Services, adapated from the UK NEA (2011), and goods provided by those services. Goods selected for valuation are highlighted in grey. (P) Provisioning, (R) Regulating, (C) Cultural, (I) Intermediate Service included in selection exercise.

Final Ecosystem Services
(P) Food
Crops
Livestock: Meat & Dairy
Game (grouse, venison)
Wild food (fungi)
(P) Fibre and fuel
Wool
Timber
Peat extraction
(P) Genetic resources
Genetic diversity of wild species
(P) Water supply
Drinking water
(R) Equable climate
C stocks in vegetation
Net C sequestration
Other GHG emissions
(R) Purification
Clean air
Clean water
(R) Hazard regulation
Reduced flooding (rivers)
Reduced flooding (coastal)
(C) Leisure, recreation, amenity
Recreational fishing
Leisure activities
Aesthetic appreciation of natural environment
Appreciation of biodiversity
(I) Intermediate services
Pollination

Selection proceeded on the basis of scale of air pollution impact, strength of evidence and valuation base, but also the requirement to attempt to quantify impacts evenly across the three categories of final services: Provisioning (P), Regulating (R) and Cultural (C). The full criteria and procedures for selection the provision of which is dependent on the provision of ecosystem services from natural capital stocks. The selection of the goods is detailed in Annex 2 (Jones et al. 2012 Annex 2).

The analysis considers the provision of the selected goods across a range of UK habitat types as appropriate to the ecosystem service[11]: enclosed farmland (arable, horticultural and improved pastures); semi-natural unimproved grassland (acidic, calcareous and neutral grasslands, bracken); woodlands (managed coniferous and broadleaved, and unmanaged); mountains moors and heathlands (bogs and dwarf shrub heaths) and freshwaters (streams, rivers and lakes). Other habitat types including urban are included where relevant for some services. The selection of the impacted habitats is detailed in Annex 2 (Jones et al. 2012 Annex 2), and was based on a reference matrix of services provided by each habitat.

The selection of final goods also implies a mix of beneficiary populations across different spatial scales and components of TEV. The market goods – livestock and timber – are assessed in terms of direct (consumptive) use value, aggregating at the national level in terms of production from agriculture and forestry sectors. Valuation of GHG emissions also implies a national beneficiary population for sequestration and mitigation benefits, based on current UK Government guidance[12]. Benefits associated with clean water (e.g. improvements in river water quality associated with reduced eutrophication of water bodies) and recreational fishing (linked to improvements in acidification) are much more sensitive to the scale of beneficiary populations. Their practical valuation highlights a number of the challenges faced in adequately accounting for the context-specific nature of non-market values associated with direct user populations of recreation sites (e.g. specialist groups such as anglers and more informal resident population users) and improvements in environmental quality at local to regional scales. The least well-defined good is that of 'appreciation of biodiversity' which is likely to incorporate both use and non-use values and can be indicative of a general preference for conservation of features of the natural environment at the local, regional and/or national scale.

3.1.3 Change in provision of the policy good

Trends in air pollutants

The historical trends in air pollutants are described in detail in Annex 1 (Jones et al. 2012 Annex 1), but are briefly summarised here. Sulphur emissions have declined by over 90% since their peak in 1970, and are projected to decline at a slower rate to 2020. Nitrogen dioxide emissions peaked around 1990, and have declined by around 50% since then, and are projected to decline still further by 2020. Ammonia emissions have declined only slightly since 1990, with little projected decline by 2020. While UK emissions of the principle precursors for ozone formation, NOx and Volatile Organic Compounds (VOCs), have declined, background ozone concentrations have been rising due to long-range transport of precursors from other countries.

Trends in pollutant deposition and non-linearities between emissions and deposition.

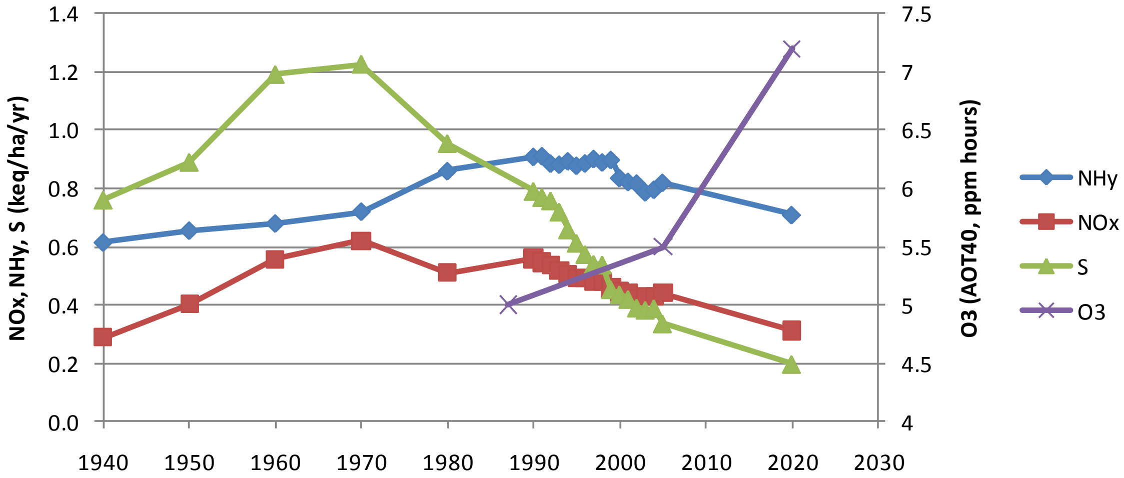
Long-term trends in deposition/concentrations of the four pollutants in this study are shown in **Figure 3.1**, and highlight the declines in sulphur and nitrogen, and the increase in mean ozone concentrations. While trends in sulphur and nitrogen deposition largely follow those of emissions, the magnitude of decline is much smaller. This is because a proportion of our air pollutants are exported abroad via long range transport (RoTAP 2010). Pollutant deposition in the UK is a function of the pollution we import, UK-based emissions and the pollution we export. As a consequence, large reductions in UK emissions do not always translate to similar reductions in deposition. This

[11] Recognising that not all habitat types are applicable to the provision of all of the final goods of interest; e.g. recreational fishing is only relevant to freshwater habitats.

[12] Damages from GHGs are determined by concentrations in the global atmosphere; i.e. the damage caused by emissions does not vary by the source of emissions, which implies a global rather than national beneficiary population from reductions in emissions. However use of current UK Government guidance for valuing GHG emissions (DECC, 2010) in practice limits the affected population to the national level. See Annex 3 for further discussion.

has particularly been the case for nitrogen dioxide over the last two decades. These complex relationships between emissions and deposition have implications for interpretation of the per-unit damage costs arising from this study where changes in deposition and therefore impact on ecosystem services may not be proportional to changes in emissions.

Figure 3.1: Trends in deposition of N in ammonia (NHy-N), N in nitrogen dioxide (NOx-N), sulphur (S) and trends in mean 6-month AOT40 ozone (O3) concentrations for the UK.



Notes: Nitrogen and Sulphur: Mean UK deposition to moorland/grassland, outputs from FRAME model, calibrated to CBED deposition in 2006, (CEH data). Ozone data are UK average 6-month AOT40 (ppmh), from monitoring network data for 2008 as a proxy for current conditions (2005), and use data for the high ozone year in 2006 as a proxy for future ozone climates likely in 2020. 1987 averages are based on data from a more limited monitoring network than available in subsequent years.

Description of scenarios

The impact of air pollution on the provision of ecosystem service – through eutrophication, acidification and direct toxicity effects – is assessed consistently across each of the seven goods of interest. Calculations are based on the specification of two emissions scenarios for NO_x, SO_x, NH_3 and O_3 emissions/concentrations:

- '***Historic emissions scenario***': based on observed emissions for the period 1987 – 2005, using 1987 as a baseline; and
- '***Projected emissions scenario***' based on forecast emissions for the period 2005 – 2020, using 2005 as a baseline.

Impacts on the provision of ecosystem services are estimated on the basis of the difference between emissions levels under each scenario and an assumed baseline. Two reference points – 1987 and 2005 – are used to specify a constant baseline level of emissions over time for each scenario. For a given year the difference in emissions is therefore:

$$\Delta emissions = observed\ or\ project\ emissions - reference\ point\ emissions \qquad [1]$$

The formulation of the baseline and emission scenarios essentially sets out two 'what if' questions for air quality policy: (i) in retrospect, what would be the difference in ecosystem service value if current levels of air quality had not been achieved; and (ii) looking forward, what is the expected impact on ecosystem service values if forecast changes in air quality are not achieved? In the context of nitrogen and sulphur, these measure the value of declines in these pollutants, in the case of ozone the scenarios measure the impact of an increase in ozone concentrations. **Table 3.2** sets out the total observed and projected emissions and deposition for nitrogen and sulphur, and AOT40 concentrations of ozone, which provided the basis for the analyses.

Both of these policy questions focus on the change in the flow of ecosystem service value over time associated with the impacts of air pollution on the market and non-market goods of interest. Detailed discussion of the scientific evidence linking air pollution, ecosystem services and the provision of these goods is provided in Annex 1 (Jones et al. 2012 Annex 1). The scale and significance of impacts is discussed in the separate summaries of the value transfer steps for each good in Annex 3 (Jones et al. 2012 Annex 3). **Figure 3.2** provides an illustration of the comparison between the two scenarios and specified baselines, in relation to average nitrogen deposition to moorland in the UK. For the historic emissions scenario, average nitrogen and sulphur deposition are based on interpolated and modelled data from the FRAME model, while for all ozone concentration trends and for nitrogen and sulphur data in the projected emissions scenario, linear trends are assumed. Modelled deposition in 2020 assumes similar changes in pollutant emissions from other European countries in response to international air pollution policy drivers.

Table 3.2: Start- and end-point pollutant emissions, deposition or concentrations used in the scenarios (nitrogen and sulphur emissions data from RoTAP (2011) and Murrells et al. (2010), ozone data provided by CEH). Future emissions derive from the UEP30 emissions scenario.

Pollutant	**1987[1]**	**2005**	**2020**
UK NOx-N emissions (ktonnes -N)	836	509	252
Imported NOx-N (ktonnes –N)	80	50	50[4]
UK NHy-N emissions (ktonnes -N)	301	251	233
Imported NHy-N (ktonnes –N)	40	40	40[4]
UK total Nitrogen **emissions** (ktonnes -N)	1137	760	485
UK total Nitrogen **deposition** (ktonnes -N)	377	319	244
Proportion of N deposition from NHy	54%	59%	66%
UK Sulphur **emissions** (ktonnes -S)	1954	319	148
Imported SOx-S (ktonnes –S)	180	59	59[4]
UK Sulphur **deposition** (ktonnes -S)	528	157	91
Proportion of acid deposition from sulphur (woodland, moorland/grassland)[2]	(28%, 35%)	(16%, 21%)	(12%, 16%)
Ozone[3] (UK average concentration, 6-month AOT40 ppmh)	5.0	5.5	7.2

[1] Nitrogen emissions and deposition data are from 1990, the earliest date available in Murrells et al. (2010) for ammonia.

[2] Both nitrogen and sulphur contribute to acid deposition. Nitrogen deposition to woodland is higher than moorland/grassland due to higher deposition velocities of ammonia.

[3] Ozone data source: see Figure 3.1 legend.

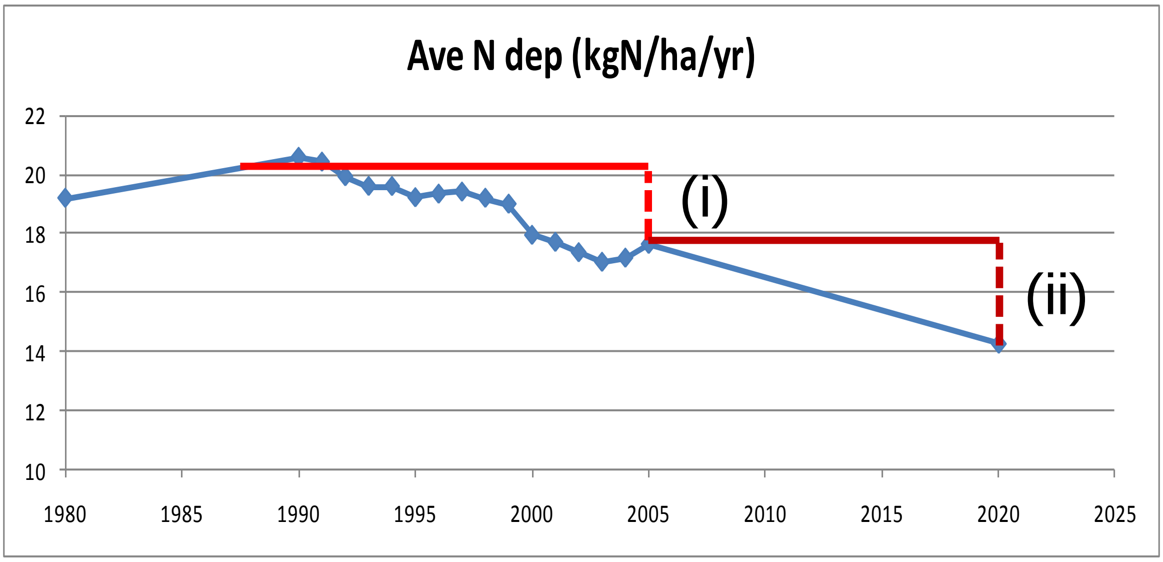
[4] Imported pollutants for 2020 assumed same as 2005, in absence of modelled data.

Assessment of ecosystem effects

Nitrogen and sulphur

For nitrogen and sulphur impacts, changes in the provision of ecosystem services are related to an average UK deposition. This uses a historical time-sequence of pollutant deposition (Fournier et al. 2004; Fowler et al. 2004) and applies unit changes in measured parameters to unit kg total N or S deposition (kg N/S per hectare per yr), or per unit acidity (keq per hectare per year) where data are available[13]. Nitrogen deposition is separately calculated to woodland and to other habitats due to differing deposition velocities of ammonia. Nitrogen and sulphur deposition from CBED deposition are available annually for most of the period 1980 to 2006, and for 2020 from the FRAME model based on projected emissions under the UEP30 energy scenario. All FRAME outputs are calibrated to CBED deposition in 2006 (CEH data). We use these deposition data as the basis for analyses in this study.

Figure 3.2: Nitrogen deposition resulting from two emissions scenarios: (i) historic emissions (1987 – 2005) and (ii) projected emissions (2005 – 2020)



Source: Mean UK nitrogen deposition to moorland/grassland. Outputs from FRAME model, calibrated to CBED deposition in 2006, (CEH data).
Notes: (i) Comparison of deposition at 1987 emissions level versus reductions achieved to present day (2005); (ii) Comparison of deposition when maintaining 2005 emissions level versus reduction to projected emissions in 2020 under UEP30 energy scenario.

Separating NO_x and NH_y effects

In most cases it is not possible to differentiate the effect of oxidised or reduced nitrogen on the receptor. There are very few scientific studies which examine these pollutants separately, and data are rarely presented in a form allowing derivation of separate response functions. Therefore, all responses are quantified for total nitrogen deposition, but resulting damage costs can be separately calculated for NO_x or NH_y according to their proportional contribution to UK deposition of total N.

[13] This does not address the spatial heterogeneity in pollutant deposition in the UK, or the spatial location of ecosystems affected, but is a necessary simplification given the scope of this study.

Ozone

Ozone data are UK averages from monitoring networks, with calculations based on a six-month accumulated ozone over a threshold concentration of 40 ppb (AOT40), expressed as ppm hours. Ozone response functions use concentration based AOT40 over the more biologically relevant Phytotoxic Ozone Dose measure PODy which takes into account the effects of climatic, plant and soil factors on the amount of ozone taken up by the plant. The differences between these methods are discussed further in Annex 1 (Jones et al. 2012 Annex 1). PODy was not used in this study simply because there is very little literature upon which to base response relationships for semi-natural systems, although PODy has been used to evaluate economic impacts on some crop species (Mills et al. 2011b). AOT40 may under-estimate ozone impacts in the UK compared with PODy (Mills et al. 2011a).

Mechanisms of impact

The air pollutants in this study impact natural systems through three principle mechanisms, eutrophication, acidification and direct toxicity. Eutrophication effects such as stimulation of plant growth are primarily due to nitrogen, but can also be due to sulphur before toxicity effects take over at high deposition loads. Acidification effects are due to nitrogen and sulphur combined, mediated through atmospheric and soil chemistry processes. Biological effects of sulphur, other than nutrient effects, are considered here under acidification (e.g. sulphur suppression of methane emissions). Direct toxicity effects at current pollutant levels occur primarily due to ozone and ammonia. In this study only ozone effects are examined, since ammonia toxicity effects are primarily localised around large point sources and require detailed spatial information on sources, concentration fields and local impact data. The eutrophication and acidification effects of ammonia are considered under the headings described above. Direct impacts from sulphur dioxide and nitrogen dioxide exposure are still possible near point sources such as very busy roads or industrial sources, but are not significant at the UK scale, given current concentrations in the UK. The mechanisms of impact are discussed in detail in Annex 1 (Jones et al. 2012 Annex 1). Lastly, it should be noted that critical loads (damage thresholds) are established based on potential impact, and where critical loads are used as the basis for valuation, e.g. for appreciation of biodiversity, current critical load exceedance represents potential future damage, not necessarily that damage has already occurred.

Timescales of recovery, and consequences for interpretation of results

Many biological systems do not respond in the same way to recovery from pollution as they do to increases in pollution. They show hysteresis, i.e. the trajectories of change and/or the timescales of change may differ, due partly to accumulation of pollutants in soils, as well as other longer-term ecological impacts. In some cases, the response functions we establish have taken into account some of these effects, based on the available scientific evidence. In other cases we know hysteresis effects may occur but have insufficient evidence to incorporate them into the modelling. These issues are dealt with on a case-by-case basis, described in detail in Annex 3 (Jones et al. 2012 Annex 3) for each valuation.

This has consequences for interpretation of some outputs from this study, for example, damage costs. Valuations for declines in nitrogen and sulphur that incorporate these hysteresis effects may produce lower damage costs than valuations for increases in those pollutants. Similarly where time lags may occur and are not incorporated, the benefits attributable to declines in pollution may be overestimated.

3.1.4 Economic valuation evidence

An overview of recent economic valuation literature relevant to the effects of air pollution on the provision of ecosystem services is provided in Annex 1 (Jones et al. 2012 Annex 1). This represents an initial 'screening' of potentially relevant studies and information, including market prices and trends for market goods and unit values estimates and value functions (e.g. willingness to pay estimates and functions) for non-market goods. More detailed assessments of selected value evidence, in line with Defra's value transfer guidelines, are presented in the supporting Annex 3 (Jones et al. 2012 Annex 3) for the valuation of each good.

3.1.5 Transfer evidence and value of the policy good

More detailed reporting of economic valuation evidence applied in the analysis is reported in Annex 3 (Jones et al. 2012 Annex 3) for the valuation of each good. Largely this is limited to use of unit values based on market prices and non-market valuation studies. In the main this reflects the broad-brush nature of the analysis, providing indicative results at the national level rather than detailed site-specific valuations which permit consideration of spatial sensitivity in economic values and other context-specific factors.

3.1.6 Aggregation

Following from the specification of the baseline and changes in the provision of ecosystem services, aggregation of estimated economic values is presented in terms of an equivalent annual value (EAV) for the historic and projected emissions scenarios. The EAV is estimated as:

$$EAV = \frac{PV}{A_{t,r}} \qquad [2]$$

Where *PV* is the present value of the change in ecosystem service value and *A* is the relevant annuity factor for time horizon *t* with discount rate *r*. The present value of the change in ecosystem service value is estimated in the standard manner:

$$PV = \sum_{t=0}^{T} \frac{V}{1+r^{t}} \qquad [3]$$

Where *V* denotes the value of the change in ecosystem service provision. Green Book guidance (HM Treasury, 2003) is followed in specifying the discount rate. Calculation of the PV of the change in ecosystem service value provides an estimate of the accumulated damage to ecosystem services from air pollution over the two scenarios, whilst the EAV provides a measure of the change in the value of the flow of ecosystem services in a given year for each scenario.

4. RESULTS

4.1 Estimated value of air pollution impacts on the provision of ecosystem services

4.1.1 Overview

Valuation of recent and projected air pollution trends on ecosystem services reveals both positive and negative impacts, differing by pollutant and by service. Impacts are summed for each pollutant, but are not summed across all pollutants for two reasons: Firstly, pollutants co-occur but have complex interactive effects, therefore combined impacts may not be equivalent to the sum of their individual effects. Secondly, while nitrogen and sulphur pollution have been declining, average ozone concentrations have been rising, and it is instructive to value these separately. Net values are presented as £million Equivalent Annual Value (EAV) with 95% Confidence Intervals (CI) in brackets.

Of the six ecosystem services selected for study, it was not possible to value impacts on all ecosystem services across all habitats, and many gaps remain. **Table 4.1** shows which services were valued in this study. Nitrogen impacts were the most comprehensively valued across the widest range of habitats, with a net benefit of £65.8 million equivalent annual value (EAV) per year due to declines in nitrogen pollution achieved since 1987, and a net benefit of £24.6 million EAV per year projected to 2020. It was only possible to comprehensively value one service for sulphur and two services for ozone, across a more limited range of habitats. Valuation of the impacts of nitrogen, sulphur and ozone are summarised in **Table 4.2**.

Table 4.1: Habitats and ecosystem services where valuation was possible

	Provisioning Services		*Regulating Services*				*Cultural Services*	
			Net GHG emissions					
	Timber production	**Livestock**	CO_2	N_2O	CH_4	**Clean water**	**Recreational fishing**	**Appreciation of biodiversity**
Nitrogen	Woodland	Improved grassland: Partially valued	Woodland, Heathlands	All semi-natural habitats	n.v.	n.v.	Upland rivers: Partially valued	Woodland, Heathland, Grasslands and Bogs.
Sulphur	n.v.	n.v.	n.v.	n.v.	Bogs	n.v.	n.v.	n.v.
Ozone	Woodland	n.v.	Woodland, Grasslands	n.v.	n.v.	n.v.	n.v.	n.v.

Notes: n.v = not valued.

Table 4.2: Summary of impact of air pollution (reductions in nitrogen and sulphur emissions, and increases in ozone concentrations) on provision of ecosystem services in Historical and Projected emissions scenarios. Values as £million Equivalent Annual Value (EAV).

	Provisioning Services		***Regulating Services***				***Cultural Services***		**Net EAV**
	Timber production	**Livestock**	**Net GHG emissions**			**Clean water**	**Recreational fishing**	**Appreciation of biodiversity**	
			CO_2	N_2O	CH_4				
Historical emissions scenario									
Reductions in Nitrogen	Loss -£1.8 (-£0.7 to -£3.5)	Loss -£4.4 (-£2.8 to -£5.7)	Loss -£21.0 (-£7.2 to -£39)	Gain £5.3 (£2.7 to £8.4)	n.v.	n.v.	Gain £0.03 (No uncertainty estimate)	*Gain* £87.7 (£13.1 to £163)*	Gain £65.8 (£5.1 - £123.2)
Reductions in Sulphur	n.v.	n.v.	n.v.	n.v.	Loss -£1.1 (-£0.4 to -£2.1)	n.v.	n.v.	n.v.	Loss -£1.1 (-£0.4 to -£2.1)
Increases in Ozone	Loss -£0.1 (-£0.07 to -£0.19	n.v.	Loss -£2.5 (-£1.6 to -£3.4)	n.v.	n.v.	n.v.	n.v.	n.v.	Loss -£2.6 (-£1.7 to -£3.6)
Projected emissions scenario									
Reductions in Nitrogen	Loss -£3.0 (-£1.6 to -£5.5)	Loss -£7.4 (-£4.7 to -£9.9)	Loss -£39.1 (-£17 to -£68)	Gain £9.3 (£4.8 to £15)	n.v.	n.v.	Gain £0.06 (No uncertainty estimate)	*Gain* £64.7 (£9.2 to £121)*	Gain £24.6 (-£9.2 to £52.7)
Reductions in Sulphur	n.v.	n.v.	n.v.	n.v.	Loss -£1.0 (-£0.3 to -£1.8)	n.v.	n.v.	n.v.	Loss -£0.9 (-£0.3 to -£1.8)
Increases in Ozone	Loss -£0.5 (-£0.3 to -£0.7)	n.v.	Loss -£10.8 (-£6.9 to -£14)	n.v.	n.v.	n.v.	n.v.	n.v.	Loss -£11.3 (-£7.2 to -£14.7)

Notes: * Estimated values for biodiversity are subject to very high levels of uncertainty (see Section A4.1.7 and Annex 3 (Jones et al. 2012c)). n.v = not valued. Lower and upper 95% confidence intervals in brackets based on uncertainty analysis.

Nitrogen

Declines in nitrogen deposition since 1987 have resulted in both positive and negative impacts on ecosystem services, summarised in **Figure 4.1**. There are negative impacts on timber production, livestock production, and carbon sequestration with a combined EAV of -£27.2m per year. There are positive impacts on nitrous oxide emissions and biodiversity, with a combined EAV of £93.0m per year. Projected further declines will have a loss of -£49.5m per year for timber, livestock and carbon sequestration, but a benefit for nitrous oxide emissions and biodiversity of £74.1m. The net EAV of nitrogen impacts across all services is an estimated benefit of £65.8m (£5.1m to £123.2m, 95% CI) and £24.6m (-£9.2m to £52.7m, 95% CI) per year for the historical and projected emissions scenarios respectively. The wide confidence intervals (which span positive and negative values) reflect the high degree of uncertainty in aggregate estimates for ecosystem services, particularly in relation to biodiversity, and results should therefore be interpreted with appropriate caution.

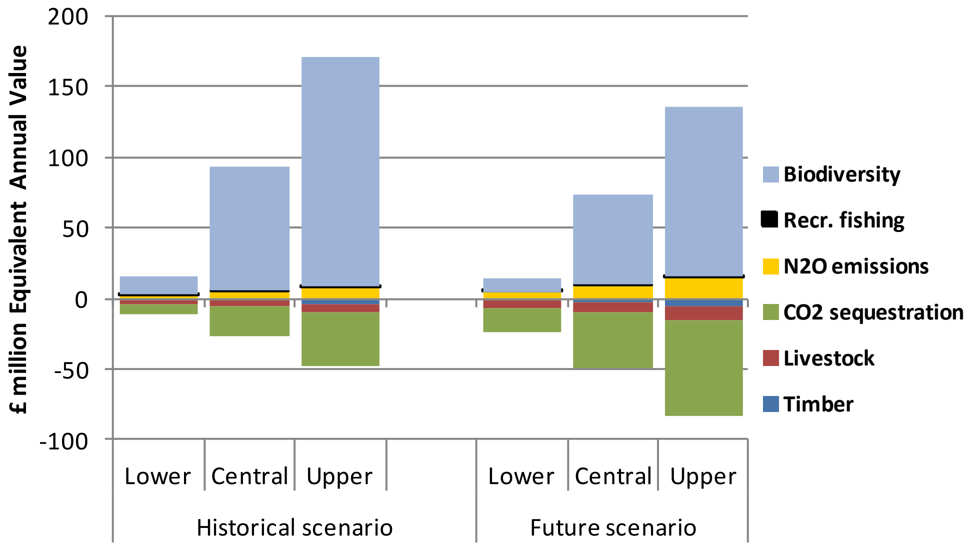


Figure 4.1: Summary of costs and benefits from reductions in nitrogen emissions (£million Equivalent Annual Value –EAV), by ecosystem service, for the Historical emissions and Projected emissions scenarios. Lower, Central and Upper represent central estimate and the lower and upper 95% confidence intervals from the uncertainty analysis.

Sulphur

Declines in sulphur deposition since 1987 have resulted in a loss of -£1.1m (-£0.4m to -£2.1m, 95% CI) per year (EAV) for methane emissions. Projected further declines will have a loss of -£0.9m (-£0.3m to -£1.8m, 95% CI) per year. We were only able to value one service in this study. However, the costs are likely to be outweighed by considerable benefits from improvements in water quality regulation and biodiversity, although we were unable to value these.

Ozone

Increases in average ozone concentrations since 1987 have resulted in a loss of -£2.6m (-£1.7m to -£3.6m, 95% CI) per year (EAV) for carbon sequestration and timber production. Projected further increases will result in a net loss of -£11.3m (-£7.2m to -£14.7m, 95% CI) per year. These estimates do not represent total impact as we were only able to value two services in this study, and were unable to use the more biologically relevant flux-based approach for quantifying impacts on these two services. Further costs may arise from damage to livestock production and to biodiversity.

A separate study (Mills et al., 2011b) has quantified ozone impacts on agricultural crop production and has predicted economic losses of £183 million per annum for a typical current year (2008) and £205 million per annum for a year representing future ozone and climatic conditions (2006).

Impacts by service

The impact varies across category of ecosystem service. For example, declines in nitrogen, at current levels of deposition, have led to reduced plant growth, with resulting negative impacts on the provisioning services and some regulating services. By contrast, the decline in nitrogen has resulted in benefits for biodiversity and some other regulating services. This leads to the potential for unintentional bias in the valuation, since provisioning services are valued by market goods which are generally easier to value, leading to an emphasis on the costs arising from declining pollution, rather than the benefits which occur largely in non-market goods which are harder to value.

Main findings are described in the following sub-sections for timber, livestock (meat and dairy production), net carbon sequestration and regulation of other GHG emissions, clean water (water quality and recreational fishing) and the appreciation of biodiversity. Step-by-step accounts of the value transfer exercise and uncertainty calculations for each good are provided in **Annex 3** (Jones et al. 2012 Annex 3). Discussion of the treatment of uncertainty is presented in Section 4.2.

4.1.2 Timber

The recent and projected changes in air quality are estimated to have a negative impact on the value of timber provided by woodlands. Reductions in nitrogen deposition are estimated to result in a loss of approximately -£1.8m and -£3.0m per year in terms of the standing stock value of timber in the UK for the historic emissions and projected emissions scenarios respectively. The increased levels of ozone are estimated to result in a loss of approximately -£0.1m and -£0.5m per year respectively for the two scenarios. These estimates account for timber produced by both the public forestry estate and privately owned-woodlands.

The analysis for timber extends the relationships established between nitrogen and ozone and tree growth in woodland habitats, which also provide the basis for valuing air pollution impacts on carbon sequestration (Section 4.3). Nitrogen acts as a nutrient stimulating tree growth, therefore declines in nitrogen deposition translate to declines in growth. Ozone on the other hand reduces tree growth therefore increases in ozone concentrations cause further declines in timber production. Sulphur and acidification effects on tree growth are not significant at the European scale, and are unlikely to be significant in the UK (See Annex 3 (Jones et al. 2012 Annex 3)). The results are interpreted in terms of indicative values that are appropriate for a broad-brush national level assessment across all woodlands in the UK. Impacts on specific woodland sites are dependent on a series of site and context-specific factors, which in general, are not accounted for in the analysis. This includes the specific management objectives of public and private woodland owners (e.g. multiple recreation, biodiversity, timber objectives) as well as factors such as the age and yield class of trees, which is assumed to average out across a national level assessment. The relationships used for nitrogen and tree growth are based on European studies which largely factor out these and other potentially confounding factors such as stand density, climate and sulphur deposition.

4.1.3 Livestock – meat and dairy production

The recent and projected changes in air quality are estimated to have a negative impact on the value of livestock production on improved grasslands (primarily beef and dairy). Reductions in nitrogen deposition are estimated to result in a loss of approximately -£4.4m and -£7.4 million per year in the historic and projected emissions scenarios respectively. This is calculated based on the very strong assumption that farmers observe the effects of changes in nitrogen input from atmospheric deposition and offset this by varying fertiliser application.

Across the historic and projected emissions scenarios, changes in input due to changes in deposition are estimated to be in the range -0.5 to 3.5 kgN per hectare per year. These estimates are around 0.1% to 2.5% of reported application rates for nitrogen fertilisers for livestock under different grazing conditions. Based on the total area of improved grassland in the UK (approximately 5.9 million hectares), the impact is judged to have a relatively insignificant impact on farm gross margins for livestock production (a reduction of approximately £1 per hectare per year), but when scaled up to the whole UK, amount to the third largest of the valued N impacts.

Impacts of nitrogen deposition on unimproved grassland livestock production (primarily sheep) have not been estimated, although reductions in atmospheric nitrogen deposition are likely to lead to reduced grassland productivity. Significant knowledge gaps are evident for relationships between N deposition, changes in grassland production and likely farm management responses (e.g. changing grazing patterns). Quantitative links between ozone and impacts on livestock production are also not available.

4.1.4 Net carbon sequestration and regulation of other GHG emissions

The recent and projected changes in air quality are estimated overall to have a negative impact on the value of GHG regulation services provided by various ecosystems and habitats. Reductions in nitrogen deposition are estimated to result in a loss of approximately -£21m and -£39.1m per year in terms of reduced carbon sequestration for the historic emissions and projected emissions scenarios while the increased levels of ozone result in a loss of approximately -£2.5m and -£10.8m per year for the two scenarios respectively. This is based on declining nitrogen deposition and increased ozone concentrations both causing reductions in above- and below-ground carbon sequestration in woodland, and reductions in below-ground carbon sequestration in heathland and grassland habitats across the UK. The valuation evidence on carbon prices applied in the analysis is sourced from UK Government guidance for valuing GHG emissions (DECC, 2010).

The effect on regulation of other GHGs (methane and nitrous oxide) varies. Reductions in sulphur deposition are estimated to result in loss of approximately -£1.1 and -£0.9m per year by reducing the suppression of methane, i.e. increasing methane emissions, from bog habitats across the UK for the projected emissions and historic scenarios. In contrast, reductions in nitrogen deposition are estimated to have a beneficial effect by reducing N_2O-N emissions from all semi-natural habitats in the UK, since emissions of N_2O are proportional to N inputs, including that from atmospheric deposition. The increase in annual value is estimated to be approximately £5.3m and £9.3m per year for the historic emissions and projected emissions scenarios respectively.

Overall the net effect of reductions in nitrogen and sulphur, and of increases in ozone on GHG regulation is judged to be negative, with the loss of value associated with reduced carbon sequestration and methane suppression outweighing the benefits of reduced nitrous oxide emissions. The reported results are based on impact relationships for nitrogen, sulphur and ozone in certain well-studied habitats, particularly woodland and heathland. However, comprehensive valuation of impacts across all habitats was not possible. Other potentially significant impacts have not been included in the analysis due to a lack of scientific evidence, or insufficient understanding of complex systems. These include the effect of nitrogen deposition on carbon sequestration in grasslands and bogs, methane emissions from semi-natural habitats other than bogs, the effect of sulphur deposition via soil acidification on carbon sequestration and nitrous oxide emissions from semi-natural habitats, and the impact of ozone on methane and nitrous oxide emissions from semi-natural habitats.

4.1.5 Clean water (river water quality)

Within the scope of this study it has not been possible to value the impact of air pollution on river water quality. The main conceptual links relate to: (i) the effect of nitrogen deposition on nitrate concentrations in freshwaters; and (ii) the effect of acid deposition (nitrogen and sulphur) on dissolved organic carbon (DOC) in acid-sensitive waters. However it is not possible at present to link changes in these individual parameters to changes in the much broader measure of ecological status of river bodies that underlies implementation of the EU Water Framework Directive (WFD)[14] or to the wider available valuation evidence for costs associated with water treatment (see Annex 3 (Jones et al. 2012 Annex 3)). For example, much WFD valuation focuses on the benefits of improving water quality from one status class to another, however the nutrient parameters used to define WFD classes focus on phosphorus and ammonia concentrations, both of which are primarily functions of sewage and agricultural inputs to freshwaters rather than atmospheric inputs. There are conceptual links between acidification and DOC concentrations, however valuation evidence on water treatment costs focuses largely on capital costs and it was not possible to obtain information on maintenance costs for this study.

4.1.6 Recreational fishing

The recent and projected changes in air pollution are estimated to have a positive impact on the value of recreational fishing trips. This finding is based on assumptions linking nitrogen deposition in upland catchments to changes in participation by anglers and associated consumer surplus, *via* changes in the nitrate concentration of rivers. The total gain in the ecosystem service value associated with recreational fishing is estimated to be approximately £0.03m and £0.06m per year in the two scenarios. Results should be interpreted as indicative values since they are based on a series of basic assumptions that are required to quantify the conceptualised impact pathway (see Annex 3 (Jones et al. 2012 Annex 3)), and other key impacts were not quantified.

It was not possible to fully quantify the impact pathway for acidification effects on fish stocks, despite considerable scientific understanding of these relationships, due to the data and resource requirements necessary to reliably upscale to the UK level. Principal limitations were representative modelling of catchment mediated acidification impacts at the UK scale, and representative data on fish population and angling use at the UK scale.

4.1.7 Appreciation of biodiversity

Recent improvements and projected changes in air quality are estimated to have a positive impact on biodiversity. This finding is based on the assumptions linking nitrogen critical load exceedance to available evidence on the value of ecosystem services provided by Biodiversity Action Plan (BAP)

[14] Directive 2000/60/EC:
http://ec.europa.eu/environment/water/water-framework/index_en.html

habitats in terms of willingness to pay for maintaining stable or increasing populations of non-charismatic BAP Priority species (trees, plants, insects excluding moths and butterflies) (Christie et al. 2010). The total gain in ecosystem service value is estimated to be approximately £65.8m and £24.6m per year for the two scenarios. It should be noted however that results are subject to very high levels of uncertainty. These are reported in Annex 3 (Jones et al. 2012 Annex 3) and should be interpreted as indicative values which are suitable in the context of this study of testing the ecosystem services approach and providing a broad-brush national level assessment, but it is not recommended that the values are used in formal project and policy analyses at this stage. Consideration should be given at the present time to the validity of the overall approach rather than refinement of specific numbers from the analysis, although subsequent work may focus on narrowing the upper and lower range bounds.

Within the timescale of this study, it was not possible to link sulphur deposition to changes in biodiversity but, in principle, this should be possible for selected habitat types, based on exceedance of acidity critical loads. There was insufficient evidence to quantitatively link ozone to species diversity loss, despite numerous studies showing impacts on the growth of individual species and changes in species balance.

4.1.8 Discussion of costs and benefits associated with changing air pollution

The results presented in **Table 4.2** and Sections 4.1.1 – 4.1.7 indicate that reductions in nitrogen and sulphur deposition, and increases in ozone can result in both gains and losses in terms of the value associated with the provision of ecosystem services. The greatest losses in value are reported for the regulating service net GHGs emissions, in particular carbon sequestration, but provisioning services are also negatively affected. The greatest gains from improvements in air quality are cultural services, specifically the appreciation of biodiversity, which outweighs the net reduction in value across all other goods, although there are high levels of uncertainty associated with these estimates. Estimated benefits with respect to recreational fishing are in general two orders of magnitude lower than other results presented in **Table 4.2**.

Both the benefits and the costs are projected to increase for most services in the projected emissions scenario, in comparison with the historical emissions scenario. It is hard to compare total costs of air pollution with other studies which have valued air pollution impacts on the environment (but see damage cost comparisons below). Smart et al. (2011) present estimated costs and benefits from a one-off 87 ktonne reduction in NH_3 emissions (their scenario 3) as £160 million benefit for human health, -£96 million cost for carbon sequestration in vegetation, £81 million benefit in reduced N_2O emissions. However, many assumptions and calculation methods differ from this study. The recent study by Mills et al. (2011b) on economic impacts of ozone on UK crops estimates total ozone damage, compared with a zero ozone reference of £183 million for 2008 (a typical current year) and £205 million for 2006 (possible conditions in 2020). The calculations in Mills et al. (2011b) take account of spatial and temporal variation in the uptake of ozone for key crops such as wheat, potato and oilseed rape which contribute 60% of the value of the 8 crops studied; AOT40-based assessments were made for the other crops. The wetter, cooler conditions in 2008 were more conducive to ozone uptake than the hotter drier conditions in 2006 and ozone impacts were almost as high in 2008 even though ozone concentrations were lower. The more sophisticated spatial analysis of impact used by Mills et al. (2011b) was not possible in this report.

4.1.9 Comparison of the two scenarios

Differences between the two scenarios are driven by a number of factors. Firstly, there are changes in the absolute level of pollutants emitted/projected to be emitted (or changes in pollutant concentrations) over the two time periods. Ozone in particular shows a steep rise in concentrations over the second time period compared with the first. Secondly, in the case of nitrogen and sulphur the ratio of emission to deposition is generally decreasing over time i.e. as our emissions decline, a greater proportion of those emissions are deposited within the UK rather than transported abroad as long-range transboundary air pollution. As the proportion of pollution we export continues to decline, in future the bulk of changes in emissions will be reflected in changes in deposition. This will lead to rising damage costs since these are calculated per unit emission. Thirdly, some relationships are non-linear, such as sulphur suppression of methane emissions, leading to differential effects as pollutant levels further decline.

4.2 Damage costs

We present indicative values for the cost or benefit per unit reduction in emissions of nitrogen dioxide and ammonia **(Table 4.3)**, and per unit increase in ozone concentrations **(Table 4.4)** in the UK. These are far from comprehensive due to the limited number of services we were able to value in this study. They are presented for comparative purposes only and it is not recommended that they are applied in formal project and policy analyses at this stage; further work is required to fill data gaps and refine assumptions that support the quantification of impact pathways.

The estimates are based on the equivalent annual value estimates reported in **Table 4.2** and the average annual difference in UK emissions (or concentrations) for each pollutant across the sequence of years used in each scenario (calculations described in Annex 3 (Jones et al. 2012 Annex 3)). Net damage costs were not calculated for SO_2, since only one service was valued and other services known to be important were not able to be valued in this study.

Note that deposition is a function of both pollutant import, changes in UK pollutant emissions and pollutant export. Calculations of damage costs in this study attributed all changes in impact to changes in UK emissions. In this study it was not possible to separately calculate the impact of deposition changes resulting solely from changes in UK emissions, but this could be one focus for future work.

Separate nitrogen damage costs for NO_x and for NH_y were calculated as follows:

- The total benefits/costs and equivalent annual value were calculated separately for the oxidised and reduced components of total N deposition for each service.
- The EAV for NO_x-N and NH_y-N was then divided by the average difference in emissions between the two emission comparisons (i.e. baseline of no change vs sequential reduction in emissions) in each year for NO_x-N and NH_y-N respectively in each scenario (see Annex 3 (Jones et al. 2012 Annex 3)).
- Damage costs per tonne NO_x-N and NH_y-N were then scaled up per tonne of NO_x and NH_y respectively, by molecular weight, assuming all NO_x is emitted as NO_2, and all NH_y is emitted as NH_3.

These calculations assume that each kg N has the same environmental impact regardless of whether it derives from oxidised or reduced N. There is increasing evidence to suggest that in some habitats NH_y is more damaging than NO_x per unit N deposited, while in other habitats the opposite may be true. However, since there is insufficient evidence at present to separately quantify response functions for oxidised and reduced N, we assume they have equal environmental impact, per unit N deposited.

In calculating separate damage costs for NO_x and NH_y impacts on biodiversity, the following assumptions have been made, following discussion with the air pollution effects research community: Calculations of exceedance are based on total N deposition and the critical load for total N, rather than assuming separate critical loads for NO_x-N and NH_y-N; the costs/benefits of changes in critical load exceedance are attributed in proportion to the separate contribution of NO_x and NH_y to total N deposition.

Damage costs for nitrogen (**Table 4.3)** are associated with declines in emissions, i.e. the ecosystem recovery component. In some cases (impacts on tree growth and carbon sequestration), the quantified relationships take into account hysteresis and time lags in recovery, therefore the associated damage costs are lower than if they were quantified for increases in these pollutants. In all other cases, the strong assumption is made that recovery occurs immediately following reductions in nitrogen deposition. Note that due to the very high level of uncertainty associated with the calculation of damage costs for biodiversity, two net damage costs are presented in Table 4.3: (a) without biodiversity; and (b) with biodiversity). Damage cost estimates for biodiversity are reported separately in Annex 3 (Jones et al. 2012 Annex 3).

Damage costs for ozone were calculated using the same principles as for nitrogen. However in contrast to nitrogen, the damage costs (**Table 4.4**) are associated with increases in mean ozone AOT40 levels. AOT40 is used by both the EU and the LRTAP Convention for risk assessment for vegetation effects in recognition that (i) effects on plants are cumulative and (ii) there is natural level of ozone detoxification, represented in this index by the 40 ppb threshold. At present, there is no accepted comparator metric for ozone which allows meaningful comparison with other pollutants considered here. In the absence of a meaningful comparator, we express damage per unit increase in UK average 6-month AOT40 (Apr – Sept), as ppm hours.

For NO_x the net damage cost estimates for impacts on ecosystem services reported here are a benefit of £77 (-£48 to £196, 95% CI) per tonne <u>reduction</u> in NO_x emissions in the historical emissions scenario, and £22 (-£97 to £129, 95% CI) per tonne <u>reduction</u> in the projected emissions scenario. For NH_3 the net damage cost estimates are a benefit of £692 (-£156 to £1,526, 95% CI) per tonne <u>reduction</u> in NH_3 emissions in the historical emissions scenario, rising to £1,246 (-£2,227 to £4,559, 95% CI) per tonne <u>reduction</u> in the projected emissions scenario. Net damage costs for ozone are a loss of -£9.1 million (-£5.8m to -£12.4m, 95% CI) per unit <u>increase</u> in 6-month AOT40 (ppm hours) in the historical scenario and -£11.4 million (-£7.3m to -£15.5m, 95% CI) per unit <u>increase</u> in the projected emissions scenario.

The indicative unit damage values for NH_3 in the future scenario are the same order of magnitude as IGCB central estimate damage costs for human health (including morbidity and mortality), calculated as £1,972 respectively per tonne pollutant emitted (IGCB, 2008, converted to 2010 prices), but are considerably lower than CAFÉ estimates which utilise different methodologies and include crop impacts as well as a slightly wider set of morbidity impacts, with values ranging from £3,827 to £9,813 per tonne for NO_x, and from £16,683 to £49,071 per tonne for NH_3 (AEA Technologies, 2005, converted to 2010 prices). The damage costs for ozone are not directly comparable with health studies. Human health impacts of ozone are calculated based on daily maximum 8-hr mean ozone concentrations (Watkiss et al. 2006), whereas calculations of ozone damage in this study are based on 6-month daylight AOT40.

The ozone damage costs can be compared with a recent crop study. The estimated unit values for ozone for two ecosystem services here are roughly one sixth of those calculated for seven key crops in the UK in 2006 and 2008, which calculated damage costs of -£60m to -£66m per unit increase in 6-month AOT40 (ppmh) (data from Mills et al. 2011b).

These are indicative damage costs and should not be regarded as comprehensive. These and damage costs from other studies have been collated in Defra report AQ0827 (Jones et al. 2014), together with robustness scores assigned based on the method of calculation. That report is a more up-to-date source of information on damage costs. Interpretation of the damage cost values presented in this study should bear in mind the following issues:

- These are net damage costs incorporating both costs and benefits associated with different ecosystem services.
- There are considerable knowledge gaps covering impacts of ozone concentrations and sulphur emissions. Calculation of sulphur damage costs will become meaningful when additional services can be valued.
- It is reasonable to expect that the indicative damage costs are 'under-estimates' due to the partial coverage of ecosystem services – as additional ecosystem services are valued, particularly regulating and cultural services where air pollution impacts are likely to be negative, they may be expected to increase.
- Damage costs associated with recovery are typically lower than those associated with pollution increase (discussed earlier in this section).
- In future (i.e. beyond 2020, the time limit of this study) we expect damage costs for nitrogen to rise further as declines in nitrogen emissions are more immediately manifest in declining deposition as the proportion of emissions that we export in the form of long-range transboundary pollution declines.

Table 4.3: Unit damage costs associated with DECLINES in nitrogen and sulphur emissions. Negative numbers represent a cost, positive numbers represent a benefit linked to declining pollution levels. Values are expressed per tonne NO_2, NH_3 or SO_2 emitted in the UK. 95% Confidence Intervals (CI) in brackets.

	Provisioning Services		*Regulating Services*				*Cultural Services*		
	Timber production	Livestock	Net GHG emissions: CO_2	Net GHG emissions: N_2O	Net GHG emissions: CH_4	Clean water	Recreational fishing	Appreciation of biodiversity	*Estimated net damage cost*
Historical emissions scenario									
Decreasing Nitrogen dioxide	-£2.6 (-£5.1 to -£1.0)	-£5.4 (-£7.1 to -£3.4)	-£30.1 (-£55.3 to -£10.2)	£6.9 (£3.5 to £10.9)	n.v.	n.v.	£0.1 (No uncertainty estimate)	£108.1 (£15.6 to £200.1)	*£77 (-£48 to £196)*
Decreasing Ammonia	-£9.7 (-£18.8 to -£3.8)	-£40.4 (-£52.9 to -£25.6)	-£124.4 (-£228.9 to -£42.2)	£42.6 (£21.3 to £66.9)	n.v.	n.v.	£0.3 (No uncertainty estimate)	£823.6 (£122.9 to £1,531.1)	*£692 (-£156 to £1,526)*
Decreasing Sulphur dioxide	n.v.	n.v.	n.v.	n.v.	-£0.7 (-£0.2 to -£1.3)	n.v.	n.v.	n.v.	*Not calculated*
Projected emissions scenario									
Decreasing Nitrogen dioxide	-£4.3 (-£8.0 to -£2.3)	-£8.8 (-£11.8 to -£5.6)	-£54.0 (-£94.0 to -£22.8)	£11.8 (£6.2 to £18.7)	n.v.	n.v.	£0.1 (No uncertainty estimate)	£77.6 (£11.1 to £141.2)	*£22 (-£97 to £129)*
Decreasing Ammonia	-£93.1 (-£170.7 to -£49.7)	-£294.1 (-£395.9 to -£186.6)	-£1,267.1 (-£2,204.0 to -£535.4)	£338.4 (£179.1 to £537.4)	n.v.	n.v.	£2.2 (No uncertainty estimate)	£2,559.4 (£364.3 to £4,792.8)	*£1,246 (-£2,227 to £4,559)*
Decreasing Sulphur dioxide	n.v.	n.v.	n.v.	n.v.	-£5.3 (-£1.6 to -£9.5)	n.v.	n.v.	n.v.	*Not calculated*

Notes: n.v = not valued. Values rounded to nearest £1, except Recreational fishing.

Table 4.4: Unit damage costs associated with INCREASES in ozone concentrations. Negative numbers represent a cost, positive numbers represent a benefit linked to changing pollution levels. Values are expressed per unit ozone AOT40, as ppm hours. Note different units to Table 4.3 above.

	Provisioning Services		***Regulating Services***				***Cultural Services***		
	Timber production	**Livestock**	**Net GHG emissions**			**Clean water**	**Recreational fishing**	**Appreciation of biodiversity**	**Net Damage Cost**
			CO_2	N_2O	CH_4				
Historical emissions scenario									
Increasing Ozone	-£476,000 (-£256,000 to -£696,000)	n.v.	-£9,580,000 (-£6,057,000 to -£13,094,000)	n.v.	n.v.	n.v.	n.v.	n.v.	-£9,104,000 (-£5,801,000 to -£12,398,000)
Projected emissions scenario									
Increasing Ozone	-£550,000 (-£297,000 to -£837,000)	n.v.	-£11,959,000 (-£7,560,000 to -£16,346,000)	n.v.	n.v.	n.v.	n.v.	n.v.	-£11,409,000 (-£7,263,000 to -£15,509,000)

Notes: n.v = not valued. Values rounded to nearest £1,000.

4.3 Sensitivity testing and uncertainties

A key challenge for this type of analysis is to ensure that uncertainties are described in a similar way across all input parameters, covering a number of different disciplines. It is notable that what may appear in some areas to be critical uncertainties may appear rather unimportant in others. The problem is increased because of the extremely broad nature of impact receptors that need to be addressed in a complete implementation of the ESA in the context of air pollution.

To address this challenge, the uncertainty analysis presented in Annex 3 (Jones et al. 2012c) provides a breakdown of each stage of the analysis, highlights the key uncertainties present in each step and defines how they are to be addressed for the services concerning timber provision, livestock productivity, CO_2 sequestration, regulation of non-CO_2 greenhouse gases, and appreciation of biodiversity.

Ranges are provided for quantifiable uncertainty, typically around ±75% of the best estimate as 95% confidence interval. One factor in the uncertainty analysis concerns exposure to pollutants (via air concentrations or deposition). It may be possible to refine the estimate of uncertainty in this area using results of Defra's model intercomparison exercise. Uncertainty in response functions is also prominent where there is a need to consider biological processes (this is not the case for all of the estimates generated, for example, effects of N deposition on livestock productivity are dealt with simply using an N-balance approach). The extent to which we have captured this uncertainty is questionable, given the limited amount of data available. For example, response functions showing effects of ozone on grassland productivity are based on a single study conducted outside the UK (Volk et al. 2006), where ozone treatments are likely to be confounded by natural variation in soil fertility (Stampfli & Fuhrer 2010).

The use of average UK pollutant deposition or concentrations in most calculations is a further source of uncertainty. The limited sensitivity analysis presented for N deposition shows that the use of average deposition rather than spatially explicit calculations can vary damage calculations in some habitats by up to 100%.

Uncertainty in the valuation step is also variable. When dealing with marketed commodities (e.g. timber, N fertiliser) it is rather small. Not surprisingly, when dealing with non-marketed goods such as appreciation of biodiversity, the uncertainty will be much larger.

An important sensitivity in many cases concerns the way that land managers (foresters, farmers, fishery owners) respond to the pressures on the systems that they manage. For example, whether they would counter a reduction in N deposition to productive land with added N fertilisation. Assumptions made here should be discussed with sector experts in more depth than has been possible in this study.

An uncertainty assessment has yet to be performed for the other services considered in this report – water quality, and recreational fishing. In these cases we conclude that questions of methodology need to be addressed before it is possible to generate a meaningful range around damage estimates.

5. CONCLUSIONS AND RECOMMENDATIONS

5.1 Ecosystems and services considered in the report

This report has addressed the quantification of effects of air pollution on a number of ecosystems (forests, livestock agriculture, arable farming, freshwater fisheries and other natural ecosystems) in terms of a variety of the provisioning, regulatory and cultural services that they offer. It is important that readers understand the context of this work. The analysis presented is not intended to provide a comprehensive overview of the effects of pollutants on ecosystems or on ecosystem services. Neither is it intended to demonstrate a full application of the methods described. The purpose of the work is instead to assess whether there is potential for making progress on an area of work that has been frequently discussed in relation to policy in the UK and elsewhere in Europe, to provide an outline illustration of methods and illustrative estimates of the magnitude of impacts.

The methods defined should be discussed more widely, for example to investigate how they can be refined to improve the quality of output and extended to give a more comprehensive overview of the effects of pollution on ecosystems. This area of work may become increasingly important in the next few years as a result of increased burdens on the environment from human activities, increased interest in the use of environmental economics, and specific policy initiatives, such as the inclusion of socio-economic analysis in the European Union's REACH legislation on chemicals.

5.2 Estimating the economic value of air pollution impacts on ecosystem services

As noted, the study estimates the economic valuation of air pollution impacts on the provision of ecosystem services in terms of a selection of market and non-market goods: timber, livestock (meat and dairy), net carbon sequestration and regulation of other GHG emissions, clean water (river water quality), recreational fishing and the appreciation of biodiversity. Two hypothetical emission scenarios are specified for NOx, SOx, NH_3 and O_3 emissions/concentrations, with changes in the provision of ecosystem services estimated on the basis of the difference between emissions levels under each scenario and an assumed baseline.

The historic emissions scenario permits an assessment of the change in ecosystem service value that would be experienced over the period 1987 - 2005 if current levels of air quality had not been achieved. The projected emission scenario assesses the potential change in ecosystem service value that will result from forecast changes in air quality over the period 2005 - 2020. The formulation of the scenarios and baselines, to which they are compared, enables the change in annual flow of ecosystem service value to be estimated.

Results reported in Section 4 highlight that changes in air quality over the periods 1987 – 2005 and 2005 – 2020 result in both gains and losses in the value of ecosystem services. Reductions in nitrogen emissions, and increases in ozone concentrations both act to reduce the value of provisioning services in terms of timber and livestock, since the productivity of these services is dependent on nitrogen inputs to woodland and improved grassland habitats, while increased ozone damages productivity. Outcomes in terms of the value of GHG regulation are more ambiguous, with both gains (reduced nitrous oxide emissions) and losses (reductions in carbon sequestration and increased methane emissions) evident. While aggregating effects across pollutants should be avoided, it is judged that losses in relation to reduced carbon sequestration and suppression of methane emissions outweigh gains associated with reduction in nitrous oxide emissions.

Improvements in air quality are estimated to have a positive effect on the value of recreational fishing and appreciation of biodiversity. The effect with respect to recreational fishing is relatively small (being at least one order of magnitude lower that other estimated impacts). Estimated values for appreciation of biodiversity are subject to high levels of uncertainty, but are much larger and potentially offset the estimated losses associated with timber, livestock and net GHG emissions. Due to evidence gaps, the value of air quality impacts on clean water (river water quality) has not been estimated.

All reported results are inevitably subject to assumptions and caveats, which stem principally from: (i) the specification of the emissions scenarios and baseline against which they are compared; (ii) the available scientific evidence and its application; (iii) the available economic value evidence and its application; (iv) frequent potential for interaction between ecosystem management practices (e.g. stocking rivers with fish, adding fertiliser to grassland or providing supplementary feed to livestock) and air pollution impacts; and (v) the lack of spatial analysis of impacts within this study. Careful interpretation of the results is therefore required, with particular recognition that a stated aim of the study is to develop and trial an ecosystem services approach. Emphasis has been placed on establishing an understanding of the potential impact pathways between air pollutant emissions and impacts on the provision of ecosystem services, and results should be interpreted as generalised and indicative, rather than representative of site-specific effects. Overall it is not recommended that the results of the study (including damage cost estimates) are applied in formal project and policy analyses at this stage; further work is required to fill data gaps and refine assumptions that support the quantification of impact pathways.

5.3 Developing a methodology for valuing air pollution impacts on ecosystem services

Application of economic valuation for policy and project analyses builds on qualitative and quantitative assessments of environmental impacts, which are informed by scientific and technical studies. Economic value estimates are context-specific and the ‘ideal’ application requires that scientific and technical assessment provide the basis for valuation by establishing the details of the good to be valued and the change in its provision; for example by documenting the baseline level provision and determining how changes in air quality will affect the quality and quantity of the provision, including the location(s) and timing of improvements, and the effects on various uses and non-market outcomes.

Recent developments in terms of the ecosystem services approach (i.e. UK NEA and TEEB) highlight the complex functioning of ecosystems and how market and non-market benefits derived by human populations are reliant on underlying ecosystem services. While to date there has been little integration of the ecosystem services approach into air quality assessments, the approach taken in this study demonstrates that it is complementary to the established impact pathway approach that has provided the basis for valuations of human health impacts and crop damages from air pollutant emissions.

Adopting an ecosystem services approach does not imply greater qualitative, quantitative and monetary evidence assessment needs than would ordinarily be expected under the impact pathway approach. The analyses highlight the need for a sound understanding and robust evidence base to enable quantification of the chain of causal relationships from the source of air pollutant emissions to changes in atmospheric air quality and subsequent impacts to receptors such as human populations, ecosystems, and buildings and materials, in order to establish the physical impacts. However, the impact pathway is longer, with causal relationships occurring *via* pollutant deposition as well as *via* concentrations in air, and often mediated via secondary ecological processes. This makes assessment more prone to gaps in knowledge or evidence.

Alternative valuation methods have been explored previously. De Nocker et al (2004) investigated what was termed a 'standard price approach', equating changes in the area subject to critical loads exceedance to the forecast costs of the Gothenburg Protocol and the National Emission Ceilings Directive. Ott et al (2007) instead used a 'repair cost' approach, interpreting the costs of repairing ecosystems as an indication of willingness to pay to protect them. Unfortunately, both approaches leave open the question of what precisely is being protected, to what degree, and why this should be of concern to society more generally. Whilst a repair cost may appear to be an expression of WTP, it leaves open the question of whether anyone is actually willing to pay for it. The results of these studies, though interesting, could thus not be easily used alongside estimates of, for example, health damage, as the underlying methodology was fundamentally different.

From a practical perspective, this study demonstrates that an 'ideal' application in the context of valuing air quality impacts on ecosystem services is an ambitious and challenging undertaking. The analysis detailed here is subject to many limiting assumptions and caveats, which arise from gaps in scientific knowledge and understanding and gaps in data availability, which would of course multiply as further goods and services are considered (see Section 5.3). However the purpose of the study is to scope out the key links between air pollutant emissions and subsequent impacts on ecosystem services. Within this context the formal frameworks offered by the impact pathway approach and the ecosystem services approach (and the two in combination) provide the necessary systematic structure that is needed to identify the links and the sources of evidence to quantify them. This is particularly important where air pollutants primarily affect the supporting services that underlie the functioning of ecosystems and subsequent impact on the provision of final goods and beneficiary populations, via eutrophication, acidification and direct toxicity, are several steps removed from these effects and are dependent on multiple processes and interactions.

Largely, the methodology applied in this study is indicative of the challenges that are faced by practical analyses that aim to address complex impacts that result from anthropogenic pressures on ecosystem service provision. In these instances the guiding principles that are established and continue to be developed by initiatives such as the UK NEA, TEEB and practical guidance documents (e.g. Defra, 2007) provide a sound basis for consistent treatment of impacts and outcomes across multiple pollutants, ecosystem services and final goods. However, in reality each assessment is context-specific and is dependent on current knowledge and evidence. The main challenges – as experienced in this study – lie in determining the level of detail that is appropriate for the analysis (i.e. trading-off a broad brush national assessment against site-specific effects), the robustness of assumptions required in light of limited data, and understanding the main sensitivities in estimated results. As with any assessment, scrutiny is required on a case-by-case basis. A particular challenge is the translation of impacts on ecological processes (often supporting services) into response functions for effects on final ecosystem services and the goods and benefits derived from them.

Overall, a key point that is reinforced by this study is that multi-disciplinary expertise is crucial to implementation of an ecosystem services approach. The assessments of the impact of air pollutants on the various final goods selected in this study cover a broad range of specialist areas, including air pollution modelling, ecology, hydrology, economics, agricultural land use management and woodland management. In specifying and developing impact pathways information and data has had to be compiled from the various disciplines to establish a coherent quantitative link to changes in the provision of ecosystem services.

5.4 Methodological and evidence gaps

With respect to the scientific evidence, various gaps in understanding of processes and impacts are discussed in Annex 1 (Jones et al. 2012 Annex 1). Key limitations are highlighted as relevant in Annex 3 (Jones et al. 2012 Annex 3) where this precludes or restricts the valuation for particular ecosystem services:

- Gaps in evidence that limit the practical specification and quantification of impact pathways are evident for all ecosystem services and final goods within the scope of this study. It was not possible to evaluate effects of all pollutants across all habitats for a single ecosystem service, with the exception of timber production, a market good derived from a single habitat type. In general the understanding and evidence related to nitrogen impacts has permitted the widest valuation for this pollutant across various semi-natural habitats. There remain considerable knowledge gaps for quantification of sulphur and ozone effects.

- The calculation of damage costs included UK and imported pollutant emissions. Modelling of impacts for 2020 was not able to take into account the future proportion of imported pollution in 2020, and assumed 2005 levels of import.

- Air pollution effects are dependent on multiple ecological processes and the complex interactions between them. The resultant impacts on the 'final' ecosystem services and goods utilised by human populations are typically several steps removed from these initial effects, increasing the potential for gaps in knowledge.

- Some potentially significant impacts have been excluded from the analysis due to an explicit lack of understanding of complex ecological systems. For example uncertainty concerning processes governing methane emissions from bogs and peat are one of the more important knowledge gaps in terms of potential impacts on UK GHG fluxes; this implies that overall judgements as to changes in UK GHG emissions could be subject to significant uncertainty.

- A number of limitations also arise through extrapolation of dose-response relationships for particular contexts (e.g. a species) to more generalised assessments at a national habitat scale. For example calculations of ozone impacts on CO_2 sequestration in grasslands are based on one potentially-confounded study conducted outside the UK. Another example is the relationship between acid deposition, freshwater Acid Neutralising Capacity (ANC) and fish numbers, where the relationships are quantifiable, but the data at the UK level are lacking for catchment acidification, fish numbers and financial value, which are required to reliably extrapolate to a national assessment.

- The broad brush approach has quantified effects based on total deposition or mean concentration for the UK and total ecosystem service value and does not consider the spatial context of pollutant impact (or valuation) across the UK.

  The analysis with respect to provisioning services (timber and livestock) highlights the need for a sufficient understanding of land use management responses to air pollution impacts. In practice these will be site-specific and based on the management objectives of land managers. Within this however there is scope to understand better the likely scale and significance of impacts (e.g. changes in nitrogen input, ozone concentration) that are required to induce management responses, and subsequent implication for the provision of final goods. The absence of this information therefore requires use of simplifying assumptions and consequently reported results need to be carefully interpreted since, crucially, they are dependent on these assumptions.

- In the case of valuing river water quality, the difficulty in tracing the impact pathway from atmospheric emissions through to changes in broad measures of the ecological status of the water environment exemplifies the practical challenges that can arise in reconciling scientific and economic valuation evidence. This is exacerbated by the multiple chemical, biological and ecological processes that determine the quality of the water environment and the initial task that is faced in attributing observed changes in quality to various influencing pressures (e.g. point source emissions and diffuse agricultural pollution). Notably this challenge is not exclusive to this study and is a wider issue with implementation of the WFD[15].

Moving to the economic valuation of air pollution impacts, the study reveals that broadly applicable valuation evidence is available for all goods of interest, covering both market and non-market values as relevant. Reported results should, however, be interpreted with due caution; the Annex 3 (Jones et al. 2012 Annex 3) commentary repeatedly emphasises that valuations are presented as indicative at the national level:

- As is reasonable to expect in a broad-brush assessment, the use of value transfer is almost entirely dependent on the basic unit value transfer approach (see Defra, 2010). However a revealing finding is that many of the valuations are based on a single point-estimate, since source studies do not report sensitivity ranges or confidence intervals. As with any practical value transfer exercise it is not possible to judge the degree of transfer error. Attempts to estimate associated uncertainties have been made as part of the overall uncertainty analysis, but require further discussion.

- Section 2.1.2 sets out the basic principles that should underlie economic valuation of ecosystem services and in particular the context-specific nature of economic values. In particular that they are dependent on the baseline provision of the good and the scale of the change in provision to be valued, and other spatially sensitive factors. In practice the scope of the analysis undertaken here does not directly address the issue that (in the main due to the above point) the available evidence is rather limited (i.e. unit values). Site specific assessments would however demand a much more rigorous treatment with respect to the spatial sensitivity of economic values and would likely encounter significant gaps in this regard from the current base of evidence.

- In a number of cases, the value transfer analysis makes strong assumptions to ensure that scientific evidence can be linked to available economic valuation evidence (e.g. appreciation of biodiversity and recreational fishing). These are explicitly stated in Annex 3 (Jones et al. 2012 Annex 3) and should be viewed as key areas for refinement of the valuation evidence base that supports this study. Of greatest importance is to refine the impact pathway linking impacts of deposition to valuation of those impacts.

- With respect to use of the estimates of damage per unit of pollutant emission (Table 4.4) in simplified impact assessments, it is important to note that they may not adequately reflect the 'true' level of damage, given non-linearities in response functions. The extent to which this is a problem for the reliability of such an impact assessment will depend on several issues: (i) the balance of costs and benefits that can be quantified adequately using linear response and valuation functions (as this determines how relevant the non-linear damages are to the decision making process); (ii) whether total effects or marginal effects are being quantified; and (iii)

[15] See, for example, eftec (2010b).

the magnitude of change considered in the impact assessment relative to the change considered in the study from which average damage costs were derived.

5.5 Recommendations

Following from Section 5.3, recommendations to address notable gaps in the evidence base relate to scientific understanding and evidence, economic valuation evidence, and/or linking science and economic evidence. Largely the required systematic frameworks (ecosystem services approach, impact pathway approach) and guidance (e.g. ecosystem services approach, value transfer) are in place, and the challenges lie in the specific details of the evidence base that need to be developed.

Scientific understanding and evidence

- **Future research**: undoubtedly future assessments will be able to refine and improve understanding on air pollution impacts on ecological functions. Several specific gaps are identified in relation to eutrophication, acidification and toxicity impacts. As a first step it is necessary to prioritise the research gaps in order that research focuses on the elements likely to be most informative, either in estimation of total damage, or method development. Interest at a European scale should be noted – a broad research programme is likely to generate the necessary information more quickly and cost-effectively than a series of separate country-specific initiatives, but should not lose sight of the need for UK-specific valuation studies.

- **Specific research gaps:** A number of areas have been identified where significant progress is readily achievable. These include: Assessment of the uncertainty in dispersion modelling and exposure to pollutant concentrations; modelling to allow reliable upscaling of catchment acidification and effects on fish populations to the national scale; improved quantification and valuation of nitrogen, ozone and acidity effects on biodiversity. Spatially explicit calculation of impact and valuation may be possible for some services.

- **Better design of research and outputs**: a challenge also exists in providing better and more appropriate scientific input for economic valuation exercises. Undertaking valuation requires good quality data particularly on the baseline and change in provision of the good and understanding links between ecological functions, better environmental quality and human wellbeing outcomes. Scientific assessments that are developed to support policy and project analyses should not be seen as isolated tasks but part of the evidence that is developed in an ecosystem services framework. This perspective will also help commissioning of research, through better understanding of where the key gaps are.

Economic valuation evidence

- **Value transfer evidence:** undoubtedly the scope for undertaking robust valuations of ecosystem service values would be considerably improved by developing further the available evidence base. This includes both market and non-market values across the range of ecosystems services. The former are typically overlooked but the UK NEA emphasises that market valuations ordinarily reflect natural capital, physical capital and human capital inputs. For provisioning services a more systematic account of the contribution of natural capital to market values would be useful. For non-market values the principles set out in the Defra value transfer guidelines apply. In particular the 'protocol for primary studies' and their reporting provides the template by which to make full use of results from new valuation studies. Where opportunities exist with respect to new studies, emphasis should also be placed on developing 'transferable, value transfer tools such that spatial sensitivity in economic values can be accounted for. For provisioning services (e.g. timber, livestock) there is particular potential to explore the use of production function models that control for the influence of various human, physical and environmental factor inputs on the production of final goods. We also recommend further work on issues pertaining to valuation of biodiversity.

- **Review of value transfer evidence for biodiversity**: The greatest uncertainty in this study is around the value transfer evidence for appreciation of biodiversity, in particular, the alignment of the critical load exceedance scenarios and the BAP scenarios of Christie et al. (2010). We suggest this is critically reviewed, with a view to whether these figures may be used in, or adapted for use in, policy appraisal. We note that subsequent work (Jones et al. 2014) has improved on this value transfer methodology, linking value transfer directly to changes in species richness.

Linking science and economics

- **Increased use of multi-disciplinary expertise:** A broad multi-disciplinary team was essential in this study. Economic valuation represents the 'final step' in the qualitative – quantitative – monetary assessment process that underlies the ecosystem services and impact pathway approaches. For a consistent evidence base to be developed, the scope of each individual component should be viewed in this wider context. This requires dialogue between policy makers, scientists and economists to establish the requirements for policy and project analyses, so that current evidence needs are fulfilled and future gaps in evidence can be addressed.

REFERENCES

AEA Technology (2005) Damages per tonne emission of PM2.5, NH3, SO2, NOx and VOCs from each EU25 Member State (excluding Cyprus) and surrounding seas http://www.cafe-cba.org/assets/marginal_damage_03-05.pdf

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

Bateman, I.J., Day, B.H., Georgiou, S. and Lake, I. (2006a) The aggregation of environmental benefit values: Welfare measures, distance decay and total WTP. Ecological Economics, 60(2): 450-460.

Bateman IJ (2007) Valuing preferences regarding environmental change. In: Pretty J, Ball A, Benton T, Guivant J, Lee D, Orr D, Pfeffer M, Ward H (eds) The SAGE handbook of environment and society. Sage, London, pp 155–171.

Bateman, I.J., Mace, G.M., Fezzi, C., Atkinson, G. And Turner, R.K. (2011) 'Economic Analysis for Ecosystem Service Assessments', Environmental and Resource Economics, 48 (2), 177-218.

CAFÉ (2005) Cost-Benefit Analysis of Policy Option Scenarios for the Clean Air for Europe (CAFE) programme, Report by AEA Technology Environment for European Commission DG Environment.

Christie, M., Hyde, T., Cooper, R., Fazey, I., Dennis, P., Warren, J., Colombo, S., and Hanley, N. (2010) Economic Valuation of the Benefits of Ecosystem Services Delivered by the UK Biodiversity Action Plan. Draft Final Report to Defra, 13th August 2010.

De Groot RS, Wilson MA, Boumans RMJ (2002) A typology for the classification, description and valuation of ecosystem functions, goods and services, special issue on the dynamics and value of ecosystem services: integrating economic and ecological perspectives. Ecol Econ 41:393–408

De Nocker, L., Vermoote, S. and Heck, T. (2004) Valuation of Environmental Impacts based on Preferences Revealed in Political Negotiations and Public Referenda. Chapter IV, NewExt Project (New Elements for the Assessment of External Costs from Energy Technologies). http://www.ier.uni-stuttgart.de/forschung/projektwebsites/newext/newext_final.pdf.

Defra (2006) An Economic Analysis to Inform the Air Quality Strategy Review Consultation. London, UK, Department for Environment, Food and Rural Affairs.

Defra (2007) Introductory Guide to Valuing Ecosystem Services, Department for Environment, Food and Rural Affairs, December 2007.

Defra (2010) Valuing Environmental Impacts: Practical Guidelines for the Use of Value Transfer in Policy and Project Appraisal, Prepared by eftec for Department for Environment, Food and Rural Affairs.

eftec (2006) Valuing Our Natural Environment, Final Report to Department for Environment, Food and Rural Affairs.

eftec (2010) Valuing Environmental Impacts: Practical Guidelines for the Use of Value Transfer in Policy and Project Appraisal, Technical Report to Department for Environment, Food and Rural Affairs.

ExternE (2005) Externalities of Energy, Methodology Update 2005, European Commission DG Research.

Fisher B, Turner RK, Zylstra M, Brouwer R, De Groot R, Farber S, Ferraro P, Green R, Hadley D, Harlow J, Jefferiss P, Kirkby C, Morling P, Mowatt S, Naidoo R, Paavola J, Strassburg B, Yu D, Balmford A (2008) Ecosystem services and economic theory: integration for policy-relevant research. Ecol Appl 18(8):2050–2067

Fournier, N., Dore, A.J., Vieno, M., Weston, K.J., Dragosits, U. & Sutton, M.A. (2004). Modelling the deposition of atmospheric oxidised nitrogen and sulphur to the United Kingdom using a multi-layer long-range transport model. Atmospheric Environment, 38, 683-694.

Fowler, D., O'Donoghue, M.O., Muller, J.B.A., Smith, R.I., Dragosits, U., Skiba, U., Sutton, M.A. & Brimblecombe, P. (2004). A chronology of nitrogen deposition in the UK between 1900 and 2000. Water, Air, & Soil Pollution: Focus, 4, 9-23.

Friedrich, R., Rabl, A., and Spadaro, J. V. (2001) Quantifying the Costs of Air Pollution: the ExternE Project of the EC, Pollution Atmosphérique, 77-104.

Hanley N, Barbier EB (2009) Pricing nature: cost-benefit analysis and environmental policy-making. Edward Elgar, London

Heal GM, Barbier EB, Boyle KJ, Covich AP, Gloss SP, Hershner CH, Hoehn JP, Pringle CM, Polasky S, Segerson K, Shrader-Frechette K (2005) Valuing ecosystem services: toward better environmental decision making. The National Academies Press, Washington

Hicks, K, Morrissey, T, Ashmore, M, Raffaelli, D, Sutton, M, Smart, J, Ramwell, C, Bealey, B and Heinemeyer A. (2008). Towards an Ecosystems Approach for Ammonia- Embedding an Ecosystem Services Framework into Air Quality Policy for Agricultural Ammonia Emissions. Defra Report NR0120.

Holland, M., Pye, S., Watkiss, P., Droste-Franke, B. and Bickel, P. (2005) Damages per tonne emission of PM2.5, NH3, SO2, NOx and VOCs from each EU25 Member State (excluding Cyprus) and surrounding seas, Report to European Commission DG Environment.

Hornung et al (1995) Effects of acidification on recreational fisheries. In: ExternE; Externalities of Energy, Volume 3 – Chapter 7: Coal and Lignite.

IGCB (2007) An Economic Analysis to inform the Air Quality Strategy Updated Third Report of the Interdepartmental Group on Costs and Benefits. http://archive.defra.gov.uk/environment/quality/air/airquality/publications/stratreview-analysis/index.htm. Accessed 1/5/2011.

IGCB (2008) Damage cost guidance. November 2008. Defra. http://archive.defra.gov.uk/environment/quality/air/airquality/panels/igcb/documents/damage-cost-calculator-guidancepaper.pdf Accessed 1/5/2011.

Jones, M.L.M., Provins, A., Harper-Simmonds, L., Holland, M., Mills, G., Hayes, F., Emmett, B.A., Hall, J., Sheppard, L.J., Smith, R., Sutton, M., Hicks, K., Ashmore, M., Haines-Young, R. (2012 Annex 1). Using the Ecosystems Services Approach to value air quality. Annex 1: Review of available evidence and outline methodology. Full technical report to Defra, project NE0117.

Jones, M.L.M., Provins, A., Harper-Simmonds, L., Holland, M., Mills, G., Hayes, F., Emmett, B.A., Hall, J., Sheppard, L.J., Smith, R., Sutton, M., Hicks, K., Ashmore, M., Haines-Young, R. (2012 Annex 2). Using the Ecosystems Services Approach to value air quality. Annex 2: Prioritisation of ecosystem services and representative ecosystems for valuation. Full technical report to Defra, project NE0117.

Jones, M.L.M., Provins, A., Harper-Simmonds, L., Holland, M., Mills, G., Hayes, F., Emmett, B.A., Hall, J., Sheppard, L.J., Smith, R., Sutton, M., Hicks, K., Ashmore, M., Haines-Young, R. (2012 Annex 3). Using the Ecosystems Services Approach to value air quality. Annex 3: Treatment of Uncertainty and Value Transfer Analysis. Full technical report to Defra, project NE0117.

Jones, L., Mills, G., Milne, A., Hayes, F., Monteith, D., Dwyer, J., Ozdemiroglu, E., Hall, J., Evans, C., Emmett, B., Sutton, M., Reis, S., Ashmore, M., Everard, M., Holland, M. (2014). Assessment of the impacts of air pollution on ecosystem services – gap filling and research recommendations. (Defra Project AQ0827), Final Report, July 2014.

Karlsson P.E., H. Pleijel, M. Belhaj, H. Danielsson, B. Dahlin, M. Andersson, M. Hansson, J. Munthe and P. Grennfelt (2005). Economic Assessment of the Negative Impacts of Ozone on Crop Yields and Forest Production. A Case Study of the Estate Östads Säteri in Southwestern Sweden. Ambio vol. 34 (2005) Nr. 1 32-40.

Mäler K-G, Aniyar S, Jansson Å (2008) Accounting for ecosystem services as a way to understand the requirements for sustainable development. PNAS 105(28):9501–9506.

Millennium Ecosystem Assessment (MEA) (2003) Ecosystems and human well-being: Synthesis. Island Press, Washington, D.C., USA.

Millennium Ecosystem Assessment (MEA) (2005) Ecosystems and human well-being: a framework for assessment. Island Press, Washington, D.C., USA.

Mills, G., Hayes, F., Simpson, D., Emberson, L., Norris, D., Harmens, H., and Büker, P. (2011a). Evidence of widespread effects of ozone on crops and (semi-)natural vegetation in Europe (1990 – 2006) in relation to AOT40 – and flux-based risk maps. Global Change Biology, 17, 592-613.

Mills, G., Hayes, F., Norris, D., Hall, J., Coyle, M., Cambridge, H., Cinderby, S., Abbott, J., Cooke, S. and Murrells, T. (2011b). Impacts of Ozone Pollution on Food Security in the UK: a Case Study for Two Contrasting Years, 2006 and 2008. Report to Defra, contract AQ0816

Murrells, T.P., Passant, N.R., Thistlethwaite, G., Wagner, A., Li, Y., Bush, T., Norris, J., Walker, C., Stewart, R.A., Tsagatakis, I., Whiting, R., Conolly, C., Okamura, S., Peirce, M., Sneddon, S., Webb, J., Thomas, J., MacCarthy, J., Choudrie, S., and Brophy, N. (2010) UK Emissions of Air Pollutants 1970 to 2008. AEA Energy & Environment.

Ott, W., Baur, M., Kaufmann, Y., Frischknecht, R. and Steiner, R. (2007) NEEDS Project: New Energy Externalities Developments for Sustainability. Deliverable D.4.2.- RS 1b/WP4. "Assessment of Biodiversity Losses". http://www.needs-project.org/RS1b/RS1b_D4.2.pdf.

ROTAP, 2010. Review Of Transboundary Air Pollution (2010). Acidification, Eutrophication, Ground Level Ozone and Heavy Metals in the UK.

Smart, J, Hicks, K, Morrissey, T, Heinemeyer A, Sutton, M, and Ashmore, M (2011). Applying the ecosystem service concept to air quality management in the UK: a case study for ammonia. Environmetrics. Pre-print online. DOI: 10.1002/env.1094

Stampfli A. & Fuhrer J. 2010. Spatial heterogeneity confounded ozone-exposure experiment in semi-natural grassland. Oecologia 162 (2): 515-522.

Turner RK, Morse-Jones S, Fisher B (2010) Ecosystem valuation: a sequential decision support system and quality assessment issues. Ann N Y Acad Sci 1185:79–101

Vlachokostas, Ch., Nastic, S. A., Achillas, Ch., Kalogeropoulos, K., Karmiris, I., Moussiopoulos, N., Chourdakis, E., Banias, G. and Limperi, N. (2010) Economic damages of ozone air pollution to crops using combined air quality and GIS modelling, Atmospheric Environment 44, 3352-3361.

Volk, M., Bungener, P., Montani, M., Contat, F. & Fuhrer, J. (2006). Grassland yield declined by a quarter in five years of free-air ozone fumigation. Global Change Biology 12, 74-83.

Watkiss, P., Holland, M., Hurley, F., Pye, S. (2006). Damage Costs for Air Pollution. AEA Technology report to Defra ED48796 Issue 4.