6 - Ecosystem services under the microscope

Ecosystem approaches can be defined as 'approaches to environmental management and policy making that aim to compare costs and benefits of management and policy options on the basis of an analysis of their impacts on the supply of benefits from ecosystems to people'.

The benefits supplied by ecosystems to people have been labelled 'ecosystem services' (MA, 2003; TEEB, 2010) and comprise such benefits as the provisioning of goods by ecosystems (e.g. wood, fish, genetic information), the regulation of environmental processes (e.g. water purification by wetlands, carbon sequestration in forests) and cultural services supplied by ecosystems (e.g. providing opportunities for recreation). The capacities of ecosystems to supply such services can be affected by air pollution and other types of environmental stress. Ecosystem degradation, for instance through air pollution, evokes a cost expressed through a reduced supply of ecosystem services, and ecosystem rehabilitation through reduction of air pollution may lead to economic benefits through enhanced ecosystem services supply.

This section focuses on ecosystem approaches applied to the field of air emissions. The potential impact of air pollution is short as well as long-range and, through reaction and transformation in the atmosphere, anthropogenic and biogenic emissions can combine to create pollutants having adverse effects on ecosystems and human health. Section 6.1 provides an overview of ecosystem approaches to environmental management. Section 6.2 examines how ecosystem approaches are being used in support of two on-going European policy formulation processes, the Gothenburg Protocol and TSAP reviews. Section 6.3 analyses methodological gaps and uncertainties and proposes means of resolving these. More detail on this section can be found in Appendix 4: Ecosystem services under the microscope.

6.1. Key concepts

6.1.1. Ecosystem services

Definition:
Ecosystem services are the goods or services provided by the ecosystem to society (MA, 2003). The supply of ecosystem services will often be variable over time, and both actual and potential future supplies of services should be included in the consideration of ecosystem services in support of environmental policy making. Ecosystems include fully natural systems, but in recent thinking also systems that are strongly modified or influenced by people, such as croplands[25].

In an Ecosystem Approach, the state or health of the ecosystem is linked to the supply of ecosystem services. Changes in ecosystem state will commonly affect ecosystem services supply, with different ecosystem services affected in a different manner.

Classification of Ecosystem services
There are three different categories of ecosystem services that are generally distinguished (MA, 2003; TEEB, 2010; EEA, 2011b): (i) provisioning services; (ii) regulation services; and (iii) cultural services. These categories are described below.

(i) Provisioning services	Reflect goods and services produced by or in the ecosystem, for example a piece of fruit or a plant with pharmaceutical properties. The goods and services may be provided by natural, semi-natural and agricultural systems and, in the calculation of the value of the service, the relevant production and harvest costs have to be considered.
(ii) Regulating services	Result from the capacity of ecosystems to regulate climate, hydrological and bio-chemical cycles, earth surface processes, and a variety of biological processes. These services often have an important spatial aspect; e.g. the flood control service of an upper watershed forest is only relevant in the flood zone downstream of the forest.
(iii) Cultural services	Relate to the non-material benefits people obtain from ecosystems through recreation, cognitive development, relaxation, and spiritual reflection. This may involve actual visits to the area, indirectly enjoying the ecosystem (e.g. through nature movies), or gaining satisfaction from the knowledge that an ecosystem containing important biodiversity or cultural monuments will be preserved.

[25] Key publications in the field are the Millennium Ecosystem Assessment, which produced a framework for analysis in 2003 and a comprehensive analysis of ecosystem services globally in 2005, and more recently the publications of the TEEB (The Economics of Ecosystems and Biodiversity) Project (e.g. TEEB, 2010).

6.1.2. Economic valuation of ecosystem services

In the cost-benefit analytical framework, valuation is normally about comparing two different investment or policy options with different environmental (and financial) implications. In other words, valuation is about understanding the societal benefits of the difference between two (or more) options, for instance two policy options involving different levels of air pollution, rather than about understanding the value of the overall stock of environmental or ecological capital.

Economic valuation methods for ecosystem services

Two types of approaches have been developed to obtain information about the value of public ecosystem services:

- **Revealed preference approach:** uses a *link with a marketed good or service* to indicate the willingness-to-pay for the service. Examples of this type of method are the travel cost method that can be used to value the recreation service provided by an area, and hedonic pricing that can be used to value environmental attributes of goods or properties sold on a market, such as a clean living environment

- **Stated preference approach:** involve soliciting people's willingness to pay for an ecosystem service or a specific environmental quality using questionnaires or choice experiments and are of particular relevance to cultural services. Non-use values, for instance those values that people may attribute to the conservation of biodiversity without any other purpose than preserving (habitat for) specific species can only be analysed with stated preference methods, however there are significant concerns regarding their validity and reliability as discussed below.

Provisioning services	Regulating services	Cultural services
Provisioning services are private goods and changes in the supply of these services can generally be **valued on the basis of observable market prices**.	The biophysical quantification may require detailed modelling, often spatially explicit (i.e. in a Geographical Information System), of the various relevant ecological and biochemical processes in an ecosystem. These services often have a public goods character and generally require non-market valuation approaches	Both stated preference approach and revealed preference approach are used, depending on the specific service.

6.1.3. Ecosystem dynamics

Ecosystem dynamics may involve irreversible and/or non-linear and/or delayed or random changes in the ecosystem as a response to ecological or human drivers. Often, ecosystem responses are subject to thresholds, as a function of feedback mechanisms intrinsic to the ecosystem. Thresholds lead to sudden and sometimes unexpected changes in ecosystems following relatively minor increases in pressure on an ecosystem. These dynamics are critically important for ecosystem approaches, also to air pollution management, because they determine the response of an ecosystem to either an increase or a reduction in pollution loads. In addition, the supply of ecosystem services is often directly linked to the state of the ecosystem. Complex dynamics are also inherent to the RAINS (Regional Air Pollution Information and Simulation) model used for modelling critical loads in relation to air pollution. RAINS comprises a number of soil chemistry-related thresholds as well as complex responses of vegetation to changing soil conditions. Exceeding thresholds may trigger a significant ecosystem response including a change in the vitality of the ecosystem and its supply of ecosystem services.

6.2. Ecosystem Approaches and Air Quality Policy

There is a need to better understand ecosystem effects before they are included in CBAs of European air policy options. Positive effects of air pollution on ecosystem services need to be accounted for, for instance nitrogen deposition will, in N limited ecosystems, generally lead to enhanced timber production and carbon sequestration.

Given that the uncertainties involved are very significant, as outlined in this chapter, there is a particular need to conduct thorough uncertainty and sensitivity analyses to indicate the robustness of the assessments before they are used in support of policy making.

Major reductions in acidification have been achieved in Europe, in particular as result of the reduction of sulphur dioxide emissions. Based on updated data on critical loads for acidification and eutrophication for Europe, it was estimated that critical loads for acidification will be exceeded at 11 percent of the European ecosystem area in 2020, compared to 34 percent in 1990 and 20 percent in 2000 (CIAM, 2007). Nitrogen deposition, from a wide variety of sources including agriculture, will however still exceed critical loads for eutrophication in 53 percent of the ecosystem area (CIAM, 2007). There is, to date, still considerable uncertainty on how these emissions and subsequent changes in ecosystems changes have affected ecosystem services supply in Europe. The aspect that has received most attention is modelling ozone damages, in particular to crops. CIAM (2007) refers to the UK-based International Cooperative Programme on Effects of Air Pollution on Natural Vegetation and Crops (ICP), that is able to detect ozone damages across 17 European countries, and which has revealed damages of ozone in, in particular, South Germany and the Mediterranean.

6.2.1. The Gothenburg Protocol

The 2007 review of the Gothenburg Protocol carried out by the Centre for Integrated Assessment Modelling, which is hosted by IIASA, (CIAM, 2007) indicated that the Protocol bases emission reduction targets on the principle of critical loads and thresholds rather than on an ecosystem approach where the benefits of reducing pollution (due to an enhanced supply of ecosystem services) are compared with the costs (in terms of pollution control measures). Nevertheless, several preliminary figures are mentioned in the review, including damage costs for ozone. No costs for nitrogen deposition were specified. The updated CIAM (2011) study analyses the cost-effectiveness of various emission reductions scenarios to improve air quality in Europe in 2020 but does not present a further specification of the economic benefits resulting from reduced pollution in ecosystems.

6.2.2. The Thematic Strategy on Air Pollution

The Commission is pursuing work aimed at quantifying benefits from reduced exposure of ecosystems to air pollution. In particular, the Commission concluded in 2007, with Consultancy Services from ARCADIS Ecolas, a road map for enabling the monetary assessment of ecosystem benefits of air pollution abatement policies (De Smet et al., 2007). This study reviewed studies aimed at valuing benefits of ecosystem responses to reduced air pollution, and found a number of constraints. First, the number of ecosystem valuation studies is limited, and there is incomplete coverage of ecosystems and services. In particular there is a lack of studies that link (reductions in) air pollution to ecosystem services and economic benefits. Second, many dose response relations are still uncertain, and third, there is lack of information on non-use values attributed to ecosystem services. Since 2006, there is an increasing amount of literature on the valuation of ecosystem services, also in a European context (TEEB, 2010). However, there are still relatively few studies that explicitly link air pollution to ecosystem benefits (e.g. Bytnerowicz et al., 2007), and new scientific efforts to elucidate this relation can be expected, among others in the context of the upcoming Horizon2020 EU research program.

6.3. Methodological gaps and uncertainties

6.3.1. Uncertainties in ecological modelling of dose-response functions

The effect of complex ecosystem dynamics need to be better understood prior to including ecosystem impacts on CBA models. In particular, there are likely to be complexities resulting from interactions between different stressors including air pollution on ecosystems. In addition, lag effects may occur, for example reducing air pollution deposition rates to below critical load levels may not immediately lead to restoration of ecosystem functioning or ecosystem services supply. These lag effects will affect the cost benefit ratio of different policy options.

Modelling the effects of air pollution on ecosystems requires modelling of the causal chain 'emissions of air pollutant -> concentrations of air pollutants -> exposure of ecosystems -> impacts on ecosystems -> changes in ecosystem services supply'. From emissions up to the step impacts on the ecosystem, the GAINS model has been applied in support of policy making. There are as yet no European models to link ecosystem changes to changes in the supply of different ecosystem services (although models for specific services exist, e.g. for impacts on timber or crop production), see e.g. Smart et al. (2011).

The science behind the responses of ecosystems to acidifying and eutrophying components is relatively well established, and there is increasing experience with modelling the impacts of ozone on vegetation. For instance, nitrogen loading and ozone exposure cause changes in plant chemistry, photosynthesis, and ecosystem carbon balance in sensitive ecosystems. As transport and deposition of emissions continues, high N loading and air pollution (especially ozone exposure) may produce similar changes in less sensitive systems. Additional responses at these and larger scales may include shifts in dominant plant species, export of nitrates and acidity to streams, rivers, and estuaries, coastal eutrophication and harmful algal blooms and, possibly, increased invasiveness by N-demanding species (Grimm et al., 2008).

Nevertheless, there remain a number of complexities in relation to ecological dose response relationships.

Ecological dose response relationships to air pollution - Remaining complexities

1 - Air pollution affecting **different organizational levels** of biological systems including individuals, communities, species, and the ecosystem. These changes are interrelated, and differ per ecosystem, as does the ecosystem's adaptive capacity to pollution loads

2 - Effect of **multiple pollutants**: where different pollutants may enhance or reduce one another's impacts, for instance by changing the resilience to another stressor

3 - Lack of information on some of the chemical, plant physiological and plant community thresholds, which are critical for understanding ecosystem change. These **thresholds**, and the lack of detailed information on their occurrence and effects, limits the validity of dose-response relations, in particular when they are extrapolated to the European scale.

4 - Pollutant-environment interactions are complicated by the fact that **biotic and abiotic factors in ecosystems change significantly over time due to natural variations** and ecological processes. Besides oscillations on a daily basis and seasonal changes, there are long-range successional developments over time periods of decades. These variations obscure the effects of pollutants and other stressors.

Furthermore, once specific changes in ecosystem state (changes in ecosystem net primary production, species composition, etc.) as a function of changes in air pollution levels are understood, these changes need to be linked to changes in the supply of ecosystem services. This step is a lot more complex for semi-natural and natural ecosystems compared to croplands, because croplands consist mostly of monocultures with a simpler vegetation composition and structure. For other ecosystems, the scientific work on air pollution impacts has focussed on forests and water bodies (e.g. CIAM, 2007). However, the relation between pollution – ecosystem change – and supply of services has not been fully established. Changes in ecosystem service supply depend on different stressors (air pollution, water pollution, resource harvest rates, etc.), are variable due to variations in ecological processes, and strongly depend on ecosystem management. Hence, it is often very difficult to single out the impact of one specific pollutant and link this to ecosystem services supply. These complexities lead to significant uncertainties that should be considered in terms of their potential implications for policy making.

6.3.2. Uncertainties in the economic analysis of ecosystem changes

There is a need to examine how marginal costs and benefits of changes in ecosystem services supply resulting from changes in air pollution can be analysed. An important question is what effect passing critical loads thresholds will have on ecosystem functioning and subsequently the supply of ecosystem services. This effect is likely to differ for different ecosystem types and different types of ecosystem services.

There is a need to better understand society's willingness to pay for biodiversity. Reducing eutrophication, in particular, may lead to lower timber production and lower carbon sequestration in nitrogen limited forest ecosystems, but may enhance biodiversity in these forests. A question is how biodiversity effects and negative impacts on other services can be compared.

Much attention has been devoted in recent decades to the development of methodologies for the valuation of ecosystem services, in particular those services not traded in the market (e.g. Daily et al., 2009). Nevertheless, there remain significant uncertainties. These relate, in particular, to the valuation of regulating and cultural ecosystem services including the habitat service. Vatn (2005) describes a number of points of general concern regarding the valuation of ecosystem services, of which two are of particular relevance in the debate on valuing ecosystem impacts of air pollution:
(i) **lack of full information on ecosystem services**; and
(ii) **Value incommensurability**. In particular, there is still a lack of studies involving the valuation of changes in the supply of specific services due to environmental change. For instance, relatively few studies have comprehensively analysed the regulating and cultural services provided by European forest ecosystems.

In view of the large diversity, in terms of ecosystem type, ecosystem use, and socio-economic and cultural setting, extrapolating values between different sites will generally lead to a low accuracy. In addition, there is likely to occur a degree of value incommensurability in the case of ecosystem services provided by European ecosystems. This means that different types of values, for instance the values related to biodiversity, cultural functions of ecosystems and values derived from products harvested in an ecosystem, cannot be measured on one and the same scale – and that different stakeholders in different countries will attach different values to ecosystem services. This is likely to be most relevant for cultural services, where values are strongly dependent on the cultural backgrounds of the people that receive the service and may depend on religious, moral, ethical and aesthetical motives. In particular, ecosystem services that supply mainly or exclusively non-use benefits such as the habitat services (biodiversity conservation) are difficult to quantify in monetary terms, and the related uncertainties in any valuation exercise for such a service are substantial.

References

References sorted by alphabet:
Abbey D. et al. (1995). Estimated long-term ambient concentrations of PM_{10} and development of respiratory symptoms in a non-smoking population. Archives of Environmental Health 50(2): 139-152. AEA, 2011. Cost Benefit Analysis for the Revision of the National Emission Ceilings Directive: Policy Options for Revisions to the Gothenburg Protocol to the UNECE Convention on Long-range Transboundary Air Pollution. European Commission framework contract ENV.C.5/FRA/2006/0071.

AEA, 2005. Methodology Paper (Volume 2) for Service Contract for carrying out cost-benefit analysis of air quality related issues, in particular in the clean air for Europe (CAFE) programme. AEAT, UK.

Bytnerowicz, A., K. Omasa, E. Paoletti, 2007. Integrated effects of air pollution and climate change on forests: A northern hemisphere perspective. Environmental Pollution 147 (2007) 438-445.

Chapman, KR, DM Mannino, JB Soriano et al., 2006. Epidemiology and costs of chronic obstructive pulmonary disease. European Respiratory Journal 2006, 27: 188–207.

Chilton, S., Covey, J., Jones-Lee, M., Loomes, G., Metcalf, H., 2004. Valuation of Health Benefits Associated with Reductions in Air Pollution. Final report. Department for Environment Food and Rural Affairs (DEFRA), May 2004.

CIAM, 2007. Background document to UNECE document 'Review of the 1999 Gothenburg Protocol', Executive Body for the Convention on Long-range Transboundary Air Pollution (2007), ECE/EB.AIR/WG.5/2007/7. Background document is prepared by the Task Force on Integrated Assessment Modelling (TFIAM) and the Centre for Integrated Assessment (CIAM).

CIAM, 2011. Scenarios for the Negotiations on the Revision of the Gothenburg Protocol under the Convention on Long-range Transboundary Air Pollution. Background paper for the 48th Session of the Working Group on Strategies and Review, Geneva, April 11-14, 2011.

CONCAWE, 2006a. Evaluation of health impacts in an environmental cost benefit analysis. CONCAWE Review Issue

CONCAWE, 2006b. Analysis of the CAFE cost benefit analysis. CONCAWE Report 4/06.

CONCAWE, 2012a. Evaluating the Value of a Life Year (VOLY). CONCAWE Review Issue 21 (2), Autumn 2012.

CONCAWE, 2012b. Cost-benefit analysis and air quality policy. CONCAWE Review Issue 21 (2), Autumn 2012.

Daily, G.C., S. Polasky, J. Goldstein, P.M Kareiva, H.A Mooney, L. Pejchar, T.H Ricketts, J. Salzman, and R. Shallenberger. 2009. Ecosystem services in decision making: time to deliver. Frontiers in Ecology and the Environment 7: 21–28.

De Smet, L., K. Devoldere and S. Vermoote, 2007. Valuation of air pollution ecosystem damage, acid rain, ozone, nitrogen and biodiversity. ENV.C.5/SER/2006/0114r. European Commission, DG ENV. 06/11867/SV. Arcadis Ecolas.

Desaigues, B., Ami, D., Bartczak, A., Braun-Kohlová, M., Chilton, S., Farreras, V., Hunt, A., Hutchison, M., Jeanrenaud, C., Kaderjak, P., Máca, V., Markiewicz, O., Metcalf, H., Navrud, S., Nielsen, J.S., Ortiz, R., Pellegrini, S., Rabl, A., Riera, R., Scasny, M., Stoeckel, M.-E., Szántó, R., Urban, J., 2007. Final Report on the Monetary Valuation of Mortality and Morbidity Risks from Air Pollution. Deliverable for WP6 of RS1b of NEEDS Project, 2007.

Desaigues, B., Ami, D., Bartczak, B., Braun-Kohlová, M., Chilton, S., Czajkowski, M., Farreras, V., Hunt, A., Hutchison, M., Jeanrenaud, C., Kaderjak, P., Máca, V., Markiewicz, O., Markowska, A., Metcalf, H., Navrud, S., Nielsen, J.S., Ortiz, R., Pellegrini, S., Rabl, A., Riera, R., Scasny, M., Stoeckel, M.-E., Szántó, R. and Urban, J. Economic valuation of air pollution mortality: A 9-country contingent valuation survey of value of a life year (VOLY). Ecological Indicators, 11 (3): 902–910. 2011.

EC4MACS, 2011. Greenhouse gases and air pollutants in the European Union: Baseline projections up to 2030. EC4MACS Interim Assessment. European Consortium for Modelling of Air Pollution and Climate Strategies.

EEA, 2011a. Revealing the costs of air pollution from industrial facilities in Europe. European Environment Agency, Copenhagen, 2011.

EEA, 2011 b. Common International Classification Ecosystem Services (CICES), 2011 Update. Paper prepared for discussion at the expert meeting on ecosystem accounts organised by the UNSD, the EEA and the World Bank, London, December 2011. European Environmental Agency, Copenhagen.

Grimm, N.B., D. Foster, P. Groffman, J.M Grove, C.S Hopkinson et al. 2008. The changing landscape: ecosystem responses to urbanization and pollution across climatic and societal gradients. Frontiers in Ecology and Environment 2008; 6: 264–272.

IIASA, 2013. Policy Scenarios for the Revision of the Thematic Strategy on Air Pollution. TSAP Report #10, Version 1.0. Editor Markus Amann, International Institute for Applied Systems Analysis IIASA. March 2013.

IIASA, 2012. Cost-benefit Analysis of Scenarios for Cost-Effective Emission Controls after 2020; Corresponding to IIASA TSAP Report #7, November 2012.

MA, 2003. Millennium Ecosystem Assessment. Ecosystems and human well-being: a framework for assessment. Report of the conceptual framework working group of the Millennium Ecosystem Assessment. Island Press, Washington, DC.

Maca, V., M. Scasny, A. Hunt, L. Anneboina and S. Navrud, 2011. Health and Environment Integrated Methodology and Toolbox for Scenario Development. Presentation of unit values for health end-points: country-specific and pooled. HEIMTSA Sixth Framework Programme. (http://www.heimtsa.eu/LinkClick.aspx?fileticket=XidEPtxYdqs%3D&tabid=2937&mid=6403&language=en-GB)

Menn P, Heinrich J, Huber RM, et al., 2012. Direct medical costs of COPD – An excess cost approach based on two population-based studies. Respiratory Medicine 106(4):540–548.

Munasinghe, M. and A. Schwab, 1993. Environmental economics and natural resource management in developing countries. World Bank, Washington, DC.

NewExt - New Elements for the Assessment of External Costs from Energy Technologies. Final Report to the European Commission, DG Research, Technological Development and Demonstration (RTD). IER (Germany), ARMINES / ENSMP (France), PSI (Switzerland), Université de Paris I (France), University of Bath (United Kingdom), VITO (Belgium). September 2004.

O'Conor R.M., and G.C. Blomquist, 1997. Measurement of consumer-patient preferences using a hybrid contingent valuation method. Journal of Health Economics Volume 16, Issue 6, December 1997, Pages 667–683.

Ostro B. (1987). Air pollution and morbidity revisited. A specification test. Journal Environmental Economics Management 10: 372-382.

Ostro B. and Rothschild S. (1989). Air pollution and acute respiratory morbidity: an observation study of multiple pollutants. Environmental Research 50: 138-247.

Priez, F. and C. Jeanrenaud, 1999. Human Costs of Chronic Bronchitis in Switzerland. Swiss Journal of Economics and Statistics. Volume (Year): 135 (1999): 287-301.

Ready, R, Navrud, S, Day, B., Dubourg R. et al., 2004. Benefit Transfer in Europe: How Reliable Are Transfers between Countries? Environmental & Resource Economics 29: 67–82, 2004.

Smart, J.C.R., K. Hicks, T. Morrissey, A, Heinemeyer, M.A. Sutton and M. Ashmore. Applying the ecosystem service concept to air quality management in the UK, a case study for ammonia. Environmetrics 2011, 22, 649-661.

Stavem, K., 2002. Association of willingness to pay with severity of chronic obstructive pulmonary disease, health status and other preference measures. International journal for Tuberculosis and Lung Diseases (6): 542–549.

TEEB, 2010. The Economics of Ecosystems and Biodiversity. Ecological and Economic Foundations. Edited by Pushpam Kumar. Earthscan, London and Washington DC.

Vatn, 2005, Vatn, A., 2005. Institutions and the Environment. Edward Elgar, Cheltenham, U.K

WHO, 2013. Review of evidence on health aspects of air pollution – REVIHAAP Project Technical Report. World Health Organisation, Regional Office for Europe.

Wilson, L. EB Devine and K. So, 2000. Direct medical costs of chronic obstructive pulmonary disease: chronic bronchitis and emphysema. Respiratory Medicine (2000) 94, 204±213.

Appendix 4: Ecosystem services under the microscope

Ecosystem Approaches in support of Policy Formulation on Air Pollution, a Review

10 December 2013

Contents

1 - Introduction

Ecosystem approaches are defined for the purpose of this Paper as 'approaches to environmental management and policy making that aim to compare costs and benefits of management and policy options on the basis of an analysis of their impacts on the supply of benefits from ecosystems to people'. The benefits supplied by ecosystems to people have been labelled 'ecosystem services' (MA, 2003) and comprise such benefits as the provisioning of goods by ecosystems (e.g. wood, fish, genetic information), the regulation of environmental processes (e.g. water purification by wetlands, carbon sequestration in forests) and cultural services supplied by ecosystems (e.g. providing opportunities for recreation). The capacities of ecosystems to supply such services can be affected by air pollution and other types of environmental stress. Hence, ecosystem degradation, for instance through air pollution, evokes a cost expressed through a reduced supply of ecosystem services, and ecosystem rehabilitation through reduction of air pollution may lead to economic benefits through enhanced ecosystem services supply.

The Paper focuses on ecosystem approaches applied to the field of air emissions. The potential impact of air pollution is long range and, through reaction and transformation in the atmosphere, anthropogenic and biogenic emissions can combine to create pollutants having adverse effects on ecosystems and human health. Increasingly these items are coupled with events linked to climate change; for example changing precipitation patterns leads to crop losses and damages as does high ozone exposure. The combined effect of these processes may be more than a simple summation depending on plant functional responses.

The specific objectives of this Technical Concept Paper are:

- to conduct a screening of the state-of-the art of ecosystem approaches;
- to examine how ecosystem approaches are being used in support of on-going European policy formulation processes – particularly the Gothenburg Protocol and TSAP reviews;
- to analyse methodological gaps and uncertainties.

The paper is structured as follows. Chapter 2 provides an overview of the state-of-the art of ecosystem approaches to environmental management, including both ecological aspects and the application of environmental economics for the analysis of ecosystem impacts. Chapter 3 examines how ecosystem approaches are being used in support of on-going European policy formulation processes, in particular the Gothenburg Protocol and TSAP reviews. Chapter 4 analyses methodological gaps and uncertainties and proposes means of resolving these. Chapter 5 presents the key policy and scientific implications of this assessment. Annex 1 provides a Glossary of main terms used in the Paper.

2 - Brief overview of ecosystem approaches to environmental management

2.1 Ecosystem services

The UN Convention on Biological Diversity has provided the following definition of an ecosystem: 'A dynamic complex of plant, animal and micro-organism communities and non-living environment interacting as a functional unit'. Ecosystem services are the goods or services provided by the ecosystem to society (MA, 2003). Their supply depends on demand from society as well as the capacity of the local ecosystem to supply the service. For example, the amount of wood extracted from an ecosystem depends on the demand for wood and the costs at which wood can be obtained. The supply of ecosystem services will often be variable over time, and both actual and potential future supplies of services should be included in the consideration of ecosystem services in support of environmental policy making.

In the last two decades, ecosystem services have emerged as a central concept in environmental management due to its potential to link the physical and economic worlds. Key publications are the Millennium Ecosystem Assessment, which produced a framework for analysis in 2003 (MA, 2003) and a comprehensive analysis of ecosystem services globally in 2005 (MA, 2005), and more recently the publications of the TEEB (The Economics of Ecosystems and Biodiversity) Project (e.g. TEEB, 2010). In the scientific literature there are nowadays over a 1500 peer reviewed studies analysing ecosystem services supply in specific ecosystems, or providing methodological support to ecosystem services analysis and modelling. A new global assessment, IPBES (Intergovernmental Panel on Biodiversity and Ecosystem Services, analogous to the IPCC) is now in an early phase and planned to become the next global assessment in this field.

Four different categories of ecosystem services are distinguished in MA (2003), which may still be the most authorative global assessment: (i) provisioning services; (ii) regulation services; (iii) cultural services; and (iv) supporting services. These categories are described below, and Table 1 presents an overview of the ecosystem services in each category. Supporting services represent the ecological processes that underlie the functioning of the ecosystem. Their inclusion in valuation may lead to double counting as their value is reflected in the other three types of services, and this category is not further addressed in this Paper, in line with TEEB (2010) and CICES (2013).

(i) Provisioning services reflect goods and services *produced* by or in the ecosystem, for example a piece of fruit or a plant with pharmaceutical properties. The goods and services may be provided by natural, semi-natural and agricultural systems and, in the calculation of the value of the service, the relevant production and harvest costs have to be considered.

(ii) Regulating services result from the capacity of ecosystems to regulate climate, hydrological and bio-chemical cycles, earth surface processes, and a variety of biological processes. These services often have an important spatial aspect; e.g. the flood control service of an upper watershed forest is only relevant in the flood zone downstream of the forest. For instance, the nursery service is classified as a regulation service. It reflects that some ecosystems provide a particularly suitable location for reproduction and involves a regulating impact of an ecosystem on the populations of other ecosystems.

(iii) Cultural services relate to the non-material benefits people obtain from ecosystems through recreation, cognitive development, relaxation, and spiritual reflection. This may involve actual visits to the area, indirectly enjoying the ecosystem (e.g. through nature movies), or gaining satisfaction from the knowledge that an ecosystem containing important biodiversity or cultural monuments will be preserved. The latter may occur without having the intention of ever visiting the area. The cultural services category also includes the habitat service that represents the benefits that people obtain from the existence of biodiversity and nature (not because biodiversity provides a number of services, but because it is important in itself).

Table 1
List of ecosystem services (based on Turner et al., 2000; MA, 2003; TEEB, 2010)

Category	Examples of goods and services provided
Provisioning services	Food Fodder (including grass from pastures) Fuel (including wood and dung) Timber, fibres and other raw materials Biochemical and medicinal resources Genetic resources Ornamentals
Regulating services	Carbon sequestration Climate regulation through control of albedo, temperature and rainfall patterns Hydrological service: regulation of the timing and volume of river flows Protection against floods by coastal or riparian systems Control of erosion and sedimentation Nursery service: regulation of species reproduction Breakdown of excess nutrients and pollution Pollination Regulation of pests and pathogens Protection against storms Protection against noise and dust Biological nitrogen fixation (BNF)
Cultural services	Habitat service: provision of a habitat for wild plant and animal species Provision of cultural, historical and religious heritage (e.g. a historical landscape or a sacred forests) Scientific and educational information Opportunities for recreation and tourism Amenity service: provision of attractive housing and living conditions

There is an increasing interest in **species richness** as an indicator for ecosystem functioning and quality, in particular in the debate on air pollution impacts on eutrophication. The general reasoning here is that eutrophication can reduce species richness in terrestrial and aquatic ecosystems by changing the nutrient availability, favouring species better adapted to high nutrient availability over species better adapted to low nutrient availability. Species richness, in other words, in an indicator for the habitat service of ecosystems, reflecting the conservation value of an ecosystem. In addition, as made explicit in the Millennium Ecosystem Assessment (2003), biodiversity including species richness is a component required for the overall functioning of an ecosystem. In general, high biodiversity increases the resilience of the ecosystem against external shocks.

In the last decades, a large number of methods to quantify species richness and/or diversity have been developed. Three main categories of indicators for species richness are briefly described below:

- **Number of species in specific classes.** Indicators presenting the species richness of an area often focus on (a combination of) specific taxonomic groups, such as mammals, meadow birds, or vascular plants. Although the number of species in specific groups is an indicator of the species diversity of an area, drawbacks are that it does not indicate the population numbers per species (which may or may not be below viable population numbers) and that it gives equal weighing to each species.

- **Biodiversity indices.** The most well-known of these indicators are the Simpson and Shannon Indices. They express the species diversity in an ecosystem, taking into account both species richness and the relative abundance of each species. However, the indicators are difficult to interpret and require a lot of data on species occurrence. In addition, they provide equal weighing to each species (attributing equal value, for instance, to a conservation flagship species and a pest).

- **Numbers of red-list and/or endemic species.** The IUCN Red List has a global cover and provides taxonomic, conservation status and distribution information on plants and animals. The number of species evaluated for the list is currently (2009) over 45,000. Certain taxonomic groups have been comprehensively assessed (e.g. mammals, birds, amphibians, freshwater crabs, warm-water reef building corals and conifers). The cover is not complete for all taxonomic groups, with data deficiencies remaining for freshwater, marine and semi-arid ecosystems. The list provides a good starting point for identifying the number of species of particular concern for nature conservation that are present in an ecosystem.

2.2 Economic valuation of ecosystem services

Valuation of ecosystem services involves a number of subsequent steps, i.e. (i) definition of the boundaries of the (eco) system and identification of the services to be studied; (ii) quantification of ecosystem services in biophysical terms; (iii) valuation of ecosystem services; and (iv) aggregation or comparison of values of different services. The services to be in or excluded from the assessment are determined by the objectives and system boundaries of the assessment. For regulating services biophysical quantification may require detailed modelling, often spatially explicit (i.e. in a GIS), of the various relevant ecological and biochemical processes in an ecosystem. Cultural services are strongly dependent on the cultural backgrounds of the people that receive the service and may depend on religious, moral, ethical and aesthetical motives. The most tangible cultural service, that can be analysed by means of a Travel Cost Valuation Method, is recreation and tourism. Other services, in particular those supplying non-use benefits (see below) are much harder to quantify and the related uncertainties in the valuation are usually substantial.

Ecosystem services can provide different types of economic value. In literature, four types of economic value are often distinguished, even though different authors have provided different classifications for these value types (e.g. Pearce and Turner, 1990; Munasinghe and Schwab, 1993; MA, 2003). Generally, the following four types are recognized: (i) direct use value; (ii) indirect use value; (iii) option value; and (iv) non-use value.

(i) Direct use value arises from the direct utilisation of ecosystems, for example through the sale or consumption of a piece of fruit. All provisioning services and some cultural services (such as recreation) have direct use value.

(ii) Indirect use value stems from the indirect utilization of ecosystems, in particular through the positive externalities that ecosystems provide. This reflects the type of benefits that regulation services provide to society.

(iii) Option value relates to risk. Because people are unsure about their future demand for a service, they are willing to pay to keep the option of using a resource in the future – insofar as they are, to some extent, risk averse. Option values may be attributed to all services supplied by an ecosystem.

(iv) Non-use value is derived from attributes inherent to the ecosystem itself. There are three types of non-use value: existence value (based on utility derived from knowing that something exists), altruistic value (based on utility derived from knowing that somebody else benefits) and bequest value (based on utility gained from future improvements in the well-being of one's descendants).

These four value types all need to be considered in the assessment of the total value of the services supplied by an ecosystem. In principle, the values are additive. Insofar as commensurable value indicators have been used, they may be summed in order to obtain the total value of the services supplied by the ecosystem. However, when analysing the ecological economics literature, there is relatively little experience with valuation of option values. The valuation of non-use values, which can only be done with stated preference methods such as Contingent Valuation Methods, is also prone to significant uncertainty. A brief overview of valuation methods is presented below.

Valuation of private goods. In the case of private goods or services traded in the market, price is the measure of marginal willingness to pay for that good, under perfect market conditions. Valuation of changes in the supply of a market ecosystem service, for instance as a function of changes in air pollution levels, requires establishing changes in the consumer and the producer surplus generated by the service. This generally requires analysing the demand and the supply curve for the ecosystem services in question. In case of price distortions, for example because of subsidies, taxes, etc., an economic (shadow) price of the good or service in question needs to be constructed. In some cases, this can be done on the basis of the world market prices following well-established approaches (Little and Scott, 1976). In case the private good is not traded in the market, because it is bartered or used for auto-consumption, shadow prices can be constructed, for instance on the basis of: (i) the costs of substitutes; or (ii) the derived benefit of the good (Munasinghe and Schwab, 1993).

Valuation of public goods. For public goods or services, the marginal willingness to pay cannot be estimated from direct observation of transactions, and the demand curves are usually difficult to construct. Two types of approaches have been developed to obtain information about the value of public ecosystem services: the revealed and the stated preference approach (Pearce and Howarth, 2000). The revealed preference approaches use a link with a marketed good or service to indicate the willingness-to-pay for the service. There are two main types of revealed preference approaches:

- *Physical linkages.* Estimates of the values of ecosystem services are obtained by determining a physical relationship between the service and something that can be measured in the market place. The main approach in this category is the damage-function (or dose-response) approach, in which the damages resulting from the reduced availability of an ecosystem service are used as an indication of the value of the service. This method can be applied to value, for instance, the hydrological service of an ecosystem.

- *Behavioural linkages*. In this case, the value of an ecosystem service is derived from linking the service to human behaviour – in particular people's expenditures to offset the lack of a service, or to obtain a service. An example of a behavioural method is the Averting Behaviour Method (ABM). There are various kinds of averting behaviour for instance defensive expenditure (a water filter) or the purchase of environmental surrogates (bottled water). The travel cost method and the hedonic pricing method are other indirect approaches using behavioural linkages.

With *stated preference* approaches, various types of questionnaires are used to reveal the willingness-to-pay of consumers for a certain ecosystem service. The most important approaches are the Contingent Valuation Method (CVM), Choice Experiments and related methods. In the last decades, CVM studies have been widely applied. It is the only valuation method that can be used to quantify the non-use values – for instance those related to biodiversity conservation - of an ecosystem in monetary terms. Various authors question their validity and reliability - both on theoretical and empirical grounds. There are two main points of criticism against CVM. First, CV estimates are sensitive to the order in which goods are valued; the sum of the values obtained for the individual components of an ecosystem is often much higher than the stated willingness-to-pay for the ecosystem as a whole. Second, CV often appears to overestimates economic values because respondents do not actually have to pay the amount they express to be willing to pay for a service (see e.g. Cummings and Harrison, 1995; Hanemann, 1995).

2.3 Ecosystem dynamics

Since some three decades, ecologists have become aware that ecosystem change is in the majority of the cases determined by complex, non-linear dynamics rather than linear responses to management and stress. These dynamics are critically important for ecosystem approaches, also to air pollution management, because they determine the response of an ecosystem to either an increase or a reduction in pollution loads. In addition, the supply of ecosystem services is often directly linked to the state of the ecosystem. Ecological research on complex dynamics is still continuous, even though the theory is now well established and increasingly integrated in environmental policy making. Note that also the RAINS (Regional Air Pollution Information and Simulation) model used for modelling critical loads in relation to air pollution is grounded in the concept of complex dynamics, as elaborated below.

In short, complex dynamics are irreversible and/or non-linear changes in the ecosystem as a response to ecological or human drivers. Below, the following key elements of complex ecosystem dynamics are briefly discussed: (i) irreversibilities; (ii) multiple states and thresholds; and (iii) stochasticity and lag-effects.

(i) Irreversible dynamics. Irreversible changes in ecosystems occur when the ecosystem is not, by itself, able to recover to its original state following a certain disturbance. Irreversible changes may be permanent, as in the global loss of a species, or they may only be reversed through substantial interventions in the ecosystem, for example in the case of reforestation on sites where natural processes would not lead to recovery of the tree cover. Irreversibility comprises different mechanisms, and can take place at different scales. For instance, it can relate to the extinction of a particular species, or the conversion of an ecosystem. It may also refer to irreversible changes in the state of an ecosystem, as in the case of a transition from a rangeland dominated by palatable grasses to one dominated by unpalatable shrubs. At the global scale, the increased loading of the atmosphere with carbon dioxide is an example of a process that can be considered as irreversible at human time scales. Irreversible change may either be rapid, involving a threshold, or more gradual. Often, it is subject to considerable uncertainty, for instance with reference to the location of the threshold, or the overall rate of change of the system following a disturbance (e.g. Scheffer and Carpenter, 2003). Note that in case of irreversible dynamics, reduction in pollution loading does not yield economic benefits since recovery of the ecosystem does not take place.

(ii) Multiple states and thresholds. Multiple states are relatively stable configurations of the ecosystem, caused by the existence of feedback mechanisms that reinforce the system to be in a particular state (Scheffer et al., 2001). The state of the ecosystem may be a consequence of physical or biological perturbation, such as changes in nutrient loading or species deletion or invasion. The probability that a disturbance leads to a shift from one state to the next depends upon the magnitude of the disturbance and on the resilience of the current state. Often, the shift between multiple states occurs suddenly and comprises the existence of threshold effects. Multiple states and thresholds have been observed in a range of ecosystems, including freshwater lakes, marine fish stocks, woodlands, rangelands, coral reefs and coastal estuaries.

A type of dynamics that occurs, in some ecosystems, in conjunction with multiple states and thresholds is hysteresis. Hysteresis occurs when the ecosystem's response to an increasing pressure follows a different trajectory from a response to a release in pressure. An example is provided by the response of an estuary to nutrient loading. At low nutrient loads, seagrass may dominate the flora, but with increased nutrient loading the phytoplankton concentrations gradually increase. At a critical load the phytoplankton concentration is so high that seagrass does not have enough light to grow. The seagrass population collapses, which allows the phytoplankton to grow to even higher concentrations. To re-establish the seagrass beds, nutrient loads have to be reduced considerably below the critical load. Other ecosystems in which hysteresis has been detected include shallow lakes, rangelands, hemlock-hardwood forests and deep lakes.

(iii) Stochasticity and lag-effects. The ecosystem may also develop as a consequence of stochastic natural conditions, for instance when ecosystem change is driven by fires or high rainfall events. In the marine environment, major changes in the dominant fish species occupying a particular niche may be triggered by relatively minor, stochastic fluctuations in the fish community. Lag effects appear when impacts of specific drivers occur with a certain delay, for example because changes need to be triggered by a specific event. For instance, in rangelands, the impact of soil degradation resulting in reduced seedling establishment may become apparent only after a fire.

Hence, complex dynamics are of major importance for the understanding of ecosystem dynamics. They determine the response of the ecosystem to management including changes in air pollutant concentration. Since the capacity of an ecosystem to supply ecosystem services depends on the state of the ecosystem, the application of ecosystem approaches to air pollution control needs to consider these complex dynamics. A summary of complex dynamics is provided in Figure 1. The left hand graph presents the traditional, but seldom applicable case of a gradual, reversible response of an ecosystem to stress (e.g. stress from air pollution). The other two graphs present different types of complex dynamics. The RAINS model uses the concept of critical loads, which is a representation of the occurrence of thresholds in soils at which a rapid decrease in pH can be expected (due to the occurrence of Calcium and Aluminium buffers), as presented in the middle graph. When the critical load (stress level) is exceeded, a rapid change in pH of the forest soil can be expected, with subsequent consequences for the vitality of the forest and the supply of ecosystem services (e.g. timber production, tourism). The right hand graph pictures an ecosystem subject to irreversible dynamics.

In relation to air pollution control, there is an increasing interest in dynamic modelling of the impacts of acid and eutrophying substances on ecosystems. Dynamic modelling provides insights in the response of ecosystems, over time, to reductions or increases in pollution loads and allows a more accurate analysis of costs and benefits of air pollution control, once these models have been developed with sufficient degree of reliability (see e.g. Hettelingh et al., 2007 for an overview) and provided that adequate monetary valuation of changes in ecosystems can be achieved.

3 - Ecosystem Approaches in Air Quality Policies

3.1 The Gothenburg Protocol

The 1999 Gothenburg (Multi-effect) Protocol is part of the Convention on Long-Range Transboundary Air Pollution (CLRTAP). The CLRTAP includes eight protocols that identify specific obligations to be taken up by the Parties and has been signed by, at present, 32 countries including most western European countries, Canada, the Russian Federation, Ukraine and the USA. The Gothenburg Protocol was signed in 1999 in Gothenburg and entered into force in 2005. It sets emissions ceilings for sulphur dioxide, nitrogen oxides, volatile organic compounds and ammonia in order to reduce acidification, eutrophication and ground-level ozone.

The Annexes of the Protocol allow Canada and the USA to participate with different commitments than other signatory parties. This is due to the different regulatory nature of Canada and the USA versus most European countries. In the EU, the Gothenburg Protocol has been implemented through the National Emission Ceilings (NEC) directive. The NEC directive is more recent than the Gothenburg Protocol and deviates slightly from it. Key environmental standards specified in the Gothenburg Protocol are listed below:

- Critical loads and levels
- Maximum allowable emissions (emission ceilings) for sulphur, nitrogen oxides (NOx), Volatile Organic Compounds (VOCs) and ammonia (NH_3).
- Emission limits for sulphur from stationary sources
- Emission limits for nitrogen oxides (NOx) from stationary sources
- Emission limits for Volatile Organic Compounds (VOCs) from stationary sources
- Emission limits for fuels and new mobile sources
- Emission limits for ammonia (NH_3) from agricultural sources

In the Gothenburg Protocol, emission limits are set for each participating country. The emission limits were negotiated on the basis of scientific assessments of pollution effects and abatement options. The selection of the specific emission levels was based on the predicted effects of the pollutants and the costs of controlling pollution. The Protocol also sets limit values for specific emission sources (e.g. combustion plants, electricity production, dry cleaning, cars and lorries) and prescribes best available techniques to be used for specific applications. Hence, the use of cost benefit approaches to establish emission targets is fundamental to the Protocol.

Central in providing the science behind the Gothenburg Protocol was the RAINS model. This model links sectoral developments and abatement measures for various pollutants with environmental impacts of air pollution. RAINS covers acidification, eutrophication, ozone damage to vegetation, and health effects due to exposure to ozone and primary and secondary particulate matter. In 2007 the RAINS model has been extended into the GAINS (Greenhouse Gas and Air Pollution Interactions and Synergies) model that also includes greenhouse gas emissions and structural measures that affect the activity levels.

Substantial amendments to the Gothenburg Protocol were agreed in May 2012. These amendments included new commitments for the reduction of $PM_{2.5}$, specific attention for black carbon as driver for both air pollution and climate change, and new commitments to reduce the emissions of sulphur dioxide, nitrogen oxides, ammonia, and volatile organic compounds. In addition, a number of new countries signed up for the Gothenburg Protocol. Significant improvements in air quality can be expected as result of the implementation of the revised Gothenburg Protocol.

Major reductions in acidification have been achieved in Europe, in particular as result of the reduction of sulphur dioxide emissions. Based on updated data on critical loads for acidification and eutrophication for Europe, it was estimated that critical loads for acidification will be exceeded at 11 percent of the European ecosystem area in 2020, compared to 34 percent in 1990 and 20 percent in 2000 (CIAM, 2007). Nitrogen deposition, from a wide variety of sources including agriculture, will however still exceed critical loads for eutrophication in 53 percent of the ecosystem area (CIAM, 2007). There is, to date, still considerable uncertainty on how these emissions and subsequent changes in ecosystems changes have affected ecosystem services supply in Europe. The aspect that has received most attention is modelling ozone damages, in particular to crops. CIAM (2007) refers to the UK-based International Cooperative Programme on Effects of Air Pollution on Natural Vegetation and Crops (ICP), that is able to detect ozone damages across 17 European countries, and which has revealed damages of ozone in, in particular, South Germany and the Mediterranean.

The CIAM (2007) review of the Gothenburg Protocol indicated that the Protocol bases emission reduction targets on the principle of critical loads and thresholds rather than on an ecosystem approach where the benefits of reducing pollution (due to an enhanced supply of ecosystem services) are compared with the costs (in terms of pollution control measures). Nevertheless, several preliminary figures are mentioned in the review, including damage costs for ozone. No costs for nitrogen deposition were specified. The updated CIAM (2011) study analyses the cost-effectiveness of various emission reductions scenario's to improve air quality in Europe in 2020 but does not present a further specification of the economic benefits resulting from reduced pollution in ecosystems.

3.2 The Thematic Strategy on Air Pollution

The Thematic Strategy on Air Pollution (TSAP) (September 2005) supplements national and preceding EU legislation by establishing objectives for air pollution and proposing measures for achieving them by 2020. The Strategy on Air Pollution is one of the seven thematic strategies provided for in the Sixth Environmental Action Programme adopted in 2002. It is the first of these strategies to be adopted formally by the Commission. It is based on research carried under by the Clean Air For Europe (CAFE) programme and the successive research framework programmes, and was adopted following a lengthy consultation process involving the European Parliament, Non-Governmental Organisations and industry and private individuals.

The TSAP covers a wide range of air quality issues and potential pollutants, with a focus on Particulate Matter. The TSAP sets health and environmental objectives and emission reduction targets for selected key pollutants. Emission reduction objectives will be delivered in stages, and aim to reduce particulate matter and ozone concentrations in air, with associated impacts on acid rain, excess nutrient nitrogen and ozone. The generation of benefits through reduced exposure of ecosystems to air pollutants is one of the drivers behind the TSAP.

Concerning ecosystem approaches, the EU legislative page states that 'there is no agreed way to assign a monetary value to ecosystem damage or the likely benefits resulting from the Strategy'. However, it is stated that 'there should be a favourable impact as a result of reducing acid rain and nutrient nitrogen inputs, resulting among other things in better protection for biodiversity'. At the same time, the Commission is pursuing work aimed at quantifying benefits from reduced exposure of ecosystems to air pollution.

In particular, the Commission concluded in 2007, with Consultancy Services from ARCADIS Ecolas, a road map for enabling the monetary assessment of ecosystem benefits of air pollution abatement policies (De Smet et al., 2007). This study reviewed studies aimed at valuing benefits of ecosystem responses to reduced air pollution, and found the following constraints:

1 - Reduction scenarios in existing studies do not match those of current European policy initiatives
2 - The number of studies is limited, and there is incomplete coverage of ecosystems and services. In particular there is a lack of studies that link (reductions in) air pollution to ecosystem services and economic benefits.
3 - The ecology of the studies was often inadequate
4 - Many dose response relations are quite uncertain
5 - There is lack of information on non-use values attributed to ecosystem services.

In terms of a pathway towards defining ecosystem approaches in support of policy making on air pollution control, the study recommends to carry out a number of case studies in representative and major EU ecosystems, in which emissions are linked to concentrations and subsequently to services and economic value of changes in services supply.

Since 2006, there still have been few studies that explicitly link air pollution to ecosystem benefits in the scientific literature (e.g. Bytnerowicz et al., 2007, see also Section 4), and it is likely that these on-going scientific advances will be reflected in the further TSAP discussions, given the stated need (De Smet, 2007) to also include ecosystem impacts in the policy formulation process.

4 - Methodological gaps and uncertainties

4.1 Overview of the assessment methodology

Applying an ecosystem services approach to air pollution policy involves expressing ecosystem impacts in monetary terms, and comparing the costs of pollution control measures to the benefits of reduced pollution loads in ecosystems. This approach is consistent with and complementary to the use of environmental cost-benefit analysis to quantify health impacts in monetary terms in support of policy making. In principle, benefits from enhanced health impacts and from enhanced supply of ecosystem services can be added in the calculation of the social welfare generated by reduced pollution in the overall environment. An important difference is that ecosystem valuation does not require the translation of health impact and mortality in monetary terms, an item that is still controversial and prone to considerable uncertainty.

The next sections of this chapter discuss the key methodological gaps and uncertainties in relation to three key steps in the assessment process that are in particular prone to uncertainty. These three key steps are: (i) establishing ecological dose-response relationship linking emissions to impacts on ecosystems; (ii) linking ecosystem change to changes in ecosystem services supply, and (iii) economic valuation of ecosystem services. The current status with regard to modelling and analysing these aspects in relation to air pollution, as well as key methodological gaps and uncertainties are analysed below. The sections will also briefly explore pathways to address the identified methodological gaps and uncertainties.

4.2 Uncertainties in ecological modelling of dose-response functions

Nitrogen loading and ozone exposure cause changes in plant chemistry, photosynthesis, and ecosystem carbon balance in sensitive ecosystems. As transport and deposition of emissions continues, high N loading and air pollution (especially ozone exposure) may produce similar changes in less sensitive systems. Additional responses at these and larger scales may include shifts in dominant plant species, export of nitrates and acidity to streams, rivers, and estuaries, coastal eutrophication and harmful algal blooms and, possibly, increased invasiveness by N-demanding species (Grimm et al., 2008). Nevertheless, there remain a number of uncertainties in relation to ecological dose response relationships. This section provides a brief overview of challenges and key uncertainties in relation to modelling interactions between air pollutants and ecosystems in the natural environment.

The most typical approach to documenting the effects of specific pollutants is a dose-response experiment, where the objective is to develop a regression equation describing the relationship between exposure and some easily measured effect (e.g., growth, yield or mortality). As analytic methods improved and ecology progressed, a broader range of effects of air pollutants is now identified and the understanding of the mechanisms of effects improved. Observations made on various temporal scales (e.g., long-term studies) and spatial scales (e.g., watershed studies) led to the recognition that air pollution can affect all organizational levels of biological systems including individuals, communities, species, and the ecosystem.

Several general points emerge from a review of ecological effects. First, air pollutants have indirect effects that are at least as important as direct toxic effects on living organisms. Indirect effects include those in which the pollutant alters the physical or chemical environment (e.g., soil properties), the plant's ability to compete for limited resources (e.g., water, light), or the plant's ability to withstand pests or pathogens. Examples are excessive availability of nitrogen, depletion of nutrient cations in the soil by acid deposition, mobilization of toxic elements such as aluminium, and changes in winter hardiness. As it is true for other complex interactions, indirect effects are more difficult to observe than direct toxic relationships between air pollutants and biota, and there may be a variety of interactions that have not yet been detected.

Damages to ecosystems are often caused by a combination of environmental stress factors. These include anthropogenic factors such as air pollution and other environmental stress factors such as low temperature, excess or limited water, and limited availability of nutrients. The specific combinations of factors differ among regions and ecosystems where declines have been observed. In addition, there is a group of substances that can be conserved in the landscape after they have been deposited in ecosystems. These substances are transformed through biotic and abiotic processes and can accumulate in the ecosystem. They include hydrogen ions ($H+$), sulphur (S), nitrogen (N), and mercury (Hg). Deposition of these pollutants can result in progressive increases in concentrations and affect ecosystems due to cumulative effects. Their effects can also continue after the stressors themselves have been reduced.

Pollutant-environment interactions are further complicated by the fact that biotic and abiotic factors in ecosystems change significantly over time due to ecological processes. Besides oscillations on a daily basis, and changes in a seasonal rhythm, there are long-range successional developments over time periods of decades. Table 2 presents an overview of key uncertainties in relation to establishing dose-response relations for air pollution impacts on ecosystems.

Table 2
Key uncertainties in dose-response relations for air pollution impacts on ecosystems
Source: EPA, 1999; CIAM, 2007; Grimms et al., 2008, Smart et al., 2011.

Uncertainty	Comment
Multiple pollutants	Pollutants interact at the ecosystem level and may enhance or alter impacts depending on the ecosystem type and the pollutants involved.
Multiple stressors	Some pollutants may reduce the resilience of the ecosystem to other pollutants or other types of stress including stress resulting from climate or land use change.
Ecosystem responses and adaptive capacities	Ecosystems poses complex dynamics where impacts in terms of pollution loading may trigger a range of positive and negative feedback mechanisms.
Ozone flux modelling	Flux-based approaches consider the uptake of O_3 by plants depending on humidity and other conditions but further work is needed.
Impacts of forests on air quality	Forest absorb $PM_{2.5}$ but there are few measurements of absorption rates in European ecosystems, and it is as yet unclear at what scale these impacts occur.

4.3 Uncertainties in linking ecosystem change to ecosystem services supply

Once specific changes in ecosystem state (e.g. changes in NPP, species composition, etc., as discussed in Section 4.2) have been modelled, they need to be linked to changes in the physical supply of ecosystem services. In this next step, there are a number of important additional methodological gaps and sources of uncertainty, as discussed in this section. These methodological gaps differ for crops and for (semi-) natural ecosystems such as forests, given that croplands are much simpler in terms of vegetation composition and structure. A brief overview of methodological gaps and uncertainties is presented below, specific for croplands and (semi-)natural ecosystems.

Croplands. Pollutants may affect processes within plants that control or alter growth and reproduction, and affect yields. Potential impacts include decreased photosynthesis, changes in carbohydrate allocation, increased foliar leaching, and increased sensitivity to stress. Air pollutants that may damage plants include O_3, SO_2, NOx and VOC. These pollutants may have direct effects on crops or may damage crops indirectly by contributing to ground level O_3 concentrations and/or acid deposition. While all of the above air pollutants may inflict stress on plants and affect crop yields, in most cases pollutants other than ozone are not a significant danger to crops (EPA, 1999). In addition, N deposition may enhance crop yields through a fertilisation effect (note that in general wet and dry deposition rates of N are substantially lower than N application rates through manure and inorganic fertilisers in most intensively managed fields).

Other ecosystem types. The work on air pollution impacts on other ecosystems has focussed on forests and water bodies. Available studies indicate the type of effects but dose response relationships have for most service not been established, the only exception being timber production from forests for which tentative data appears to be available (CIAM, 2007). The dose response relations for other ecosystems are substantially more complex than for cropland due to a number of factors: (i) the large diversity of ecosystems and their soil types, vegetation, etc.; (ii) the diversity of services provided by ecosystems; and (iii) in addition to ozone, some ecosystems are also significantly affected by SO_2 and nitrogen deposition. Table 3 provides an overview of the current information available on the impacts of air pollution on ecosystem services supply.

Table 3
Air pollution impacts on ecosystem services substantiated in the literature.
Adapted from EPA (1999), Bytnerowicz et al. (2007), Grimm et al. (2008) and Smart et al. (2011).

Ecosystem service	Impacts
Provisioning services	
Timber production	Loss of timber production due to ozone damage and acidification. Positive effect of nitrogen fertilisation can occur in nitrogen limited ecosystems.
Fish production	Acidification may lead to reduced fish production in lakes. Effects of eutrophication on fish populations may be positive (productivity) or negative (algal blooms, changes in species composition)
Regulating services	
Carbon sequestration	Positive impacts from nitrogen deposition in nitrogen limited ecosystems, negative impacts from ozone pollution and acidification.
Cultural services	
Recreation	Reduced recreation in ecosystems visually affected by acidification or eutrophication, reduced opportunities for recreational fishing (e.g. salmon fishing) in acidified freshwater
Biodiversity conservation	Changes in biodiversity and rare species throughout Europe due to eutrophication and acidification.

It is clear that there are considerable uncertainties related to linking air pollution to impacts on the physical supply of ecosystem services, in particular for other services than crop production. In general, such relations can only be established with elaborate models incorporating drivers for ecosystem change, key ecosystem state variables, and services supply. Contrary to effects on crop yields, it is generally very difficult to establish such relations with experiments because it is next to impossible to expose an overall ecosystem to controlled pollutant concentrations and because analysing ecosystem service supply under different pollutant concentrations based on observations faces the challenge of singling out the impact of the pollutant in between a myriad of other factors driving ecosystem services supply.

Another important factor is that ecosystem changes, for instance effects of acidification are subject to complex dynamics. Non-linear responses including a strong response once a threshold is passed can be expected for impacts of acidification and eutrophication (Scheffer et al., 2001) and potentially in ecosystems also in relation to ozone exposure[31]. In general, for many ecosystems, based on currently available information, it appears as if ecosystem services supply is strongly related to ecosystem state rather than the amount of pressure exerted on an ecosystem. In other words, the deposition of acidifying or eutrophying substances in an ecosystem only has a strong impact on the supply of ecosystem services once a threshold is passed (e.g. a buffer is exceeded) and the ecosystem shifts to an alternative state. This further complicates the analysis of ecosystem impacts, since the role of thresholds is crucial, and there remains uncertainty on the precise pressure at which the threshold is exceeded.

Hence, the key methodological gaps and sources of uncertainty in establishing the relation between ecosystem change and services supply are as follows. First, there are insufficient studies to establish general dose response relations, even for those ecosystem services for which there are concrete studies available. Second, changes in ecosystem services are guided by the complex dynamics of ecosystems, and there is insufficient information on thresholds determining ecosystem responses to acidification and eutrophication. Third, there is insufficient insight in how the effects of different pollutants may influence one another, and how these effects may interact with other pressures on ecosystems, for instance from climate change. These key sources of uncertainty need to be addressed in further scientific assessments before an ecosystem services approach can be quantitatively applied in support of policy making in the field of air pollution control.

[31] Holland et al. (2002) assume linear dose response relations for crops, but this linearity has not been confirmed for ecosystem responses to ozone.

4.4 Uncertainties in the economic analysis of ecosystem changes

Applying ecosystem approaches in support of policy formulation requires analyses of the economic costs or benefits of a change in ecosystem services supply due to changes in air pollution. Much attention has been devoted in recent decades to the development of methodologies for the valuation of ecosystem services, in particular those services not traded in the market (e.g. Daily et al., 2009).

In spite of these advances, there are still important uncertainties remaining in the field of ecosystem services valuation. Vatn (2005) describes the following four main points of general concern regarding the valuation of ecosystem services (i) a lack of full information on ecosystem services; (ii) value incommensurability; (iii) the problem of composition; and (iv) the income-dependency of willingness to pay estimates. A brief overview of these points is provided below.

(i) A lack of full information on ecosystem services. A lack of information is a frequent constraint to ecosystem valuation. For instance, there may be only approximate indication of the actual use level of the service, its marginal value in case of strong changes in supply, ecosystem dynamics and how they influence future supply of the service, etc. These constraints progressively increase at coarser scales and with increasing complexity of the ecosystem. In particular, at the European scale there may be a lack of data on the response of specific ecosystem types to pollution levels, and there may be unknown price effects related to changes in ecosystem services supply (for instance changes in crop damages).

(ii) Value incommensurability. Value incommensurability means that different types of values, for instance the values related to biodiversity, cultural functions of ecosystems and values derived from products harvested in an ecosystem, cannot be measured on one and the same scale. This argument is based on the observation that individuals have different motives for managing ecosystems, and that they therefore have difficulty in interpreting services and values along one dimension – as in the case of comparing positive effects on biodiversity with negative effects on timber production.

(iii) The problem of composition. The problem of composition indicates that the supply of an ecosystem service is always dependent on the functioning of the ecosystem supplying the service, and that demarcating parts of the environment for the purpose of valuation may lead to underestimation of the value of the ecosystem at large.

(iv) The income-dependency of willingness-to-pay estimates. The income-dependency of willingness-to-pay (WTP) estimates is a concern where there are large income discrepancies between different stakeholders. The WTP estimate is bound by the income of the respondent and restricts the articulation of unrealistically high WTP statements in a contingent valuation study (Arrow et al., 1993). Nevertheless, the wide range of WTP estimates for avoiding health impacts, as also indicated by the differences between median and average values in this regard, indicates the potential magnitude of the uncertainties involved. It can be expected that uncertainties will be comparably high for WTP estimates related to ecosystem impacts (e.g. on species richness).

There is increasing experience with the valuation of some ecosystem services, in particular provisioning services and several of the regulating services such as water regulation and carbon sequestration as well as recreation and tourism. Increasingly, the outcomes of such studies are used in the policy formulation process, including in the EU (TEEB, 2010). Nevertheless, there are still important methodological challenges remaining to the valuation of a whole range of ecosystem services. For the services affected by air pollution in Europe, these challenges are particularly relevant with regards to the impacts on biodiversity, and it is questionable if impacts of air pollution on this service can be meaningfully valued in economic terms. For carbon sequestration, there is a factor 2 to 3 uncertainty on the appropriate price to use for a unit of carbon, on the basis of a comparison of the different estimates for the marginal social damage costs of carbon, the prices for which carbon is traded in the carbon market, and the prices included in governmental guidelines for Environmental CBA. For recreation, there are currently insufficient data to establish the general relation between a decline in the recreational quality and the loss of economic value in particular for forests. For lakes, several case studies have been done on willingness to pay to avoid a reduction in water quality, but insufficient to establish a relation that is applicable at a wider, let alone European, scale. Hence, further advances in valuation methodologies are required before European wide economic analyses of the effects of changes in air pollution levels can be conducted.

5 - Policy and Scientific Implications

There are currently still relatively few studies that quantify impacts of air pollution on ecosystem services supply in monetary terms. For example, Holland et al. (2002) analyse impacts of ozone on crop production, CIAM (2007) includes work on the impacts of ozone on timber production, and Smart et al. (2011) analyse the costs of ammonia leading to the release of greenhouse gas in the UK. Based on a review of the current understanding of ecosystem services modelling and valuation in the context of air pollution, CONCAWE recommends the following:

1 - There is a need to better understand ecosystem effects before they are included in CBAs of European air policy options. Positive effects of air pollution on ecosystem services need to be accounted for, for instance nitrogen deposition will, in N limited ecosystems, generally lead to enhanced supply of ecosystem services such as timber production and carbon sequestration.

2 - There is a need to examine how marginal costs and benefits of changes in ecosystem services supply resulting from changes in air pollution can be analysed. An important question is what effect passing critical loads thresholds will have on ecosystem functioning and subsequently on the supply of ecosystem services. This effect is likely to differ for different ecosystem types and different types of ecosystem services.

3 - The effects of complex ecosystem dynamics need to be better understood prior to including ecosystem impacts on CBA models. In particular, there are likely to be complexities resulting from interactions between the multiple stressors on ecosystems including air pollution. In addition, there may be lag effects occurring, for example reducing air pollution deposition rates to below critical load levels may not immediately lead to restoration of ecosystem functioning or ecosystem services supply. These lag effects will affect the cost-benefit ratio of different policy options.

4 - There is a need to better understand society's willingness to pay for biodiversity. Reducing eutrophication, in particular, may lead to lower timber production and carbon sequestration in nitrogen limited forest ecosystems, but may enhance biodiversity in these forests. A question is how biodiversity effects and negative impacts on other services can be compared.

5 - Given that the uncertainties involved are very significant, as outlined in this Appendix, there is a particular need to conduct thorough uncertainty and sensitivity analyses to indicate the robustness of the assessments before they are used in support of policy making.

References

Arrow, K., R. Solow, P.R. Portney, E.E. Learner, R. Radner, H. Schuman (1993), Report of the NOAA Panel on Contingent Valuation. Federal Reg. 58, Washington, DC

Bytnerowicz, A., K. Omasa, E. Paoletti, 2007. Integrated effects of air pollution and climate change on forests: A northern hemisphere perspective. Environmental Pollution 147 (2007) 438-445.

CIAM, 2007. Background document to UNECE document 'Review of the 1999 Gothenburg Protocol', Executive Body for the Convention on Long-range Transboundary Air Pollution (2007), ECE/EB.AIR/WG.5/2007/7. Background document is prepared by the Task Force on Integrated Assessment Modelling (TFIAM) and the Centre for Integrated Assessment (CIAM).

CIAM, 2011. Scenarios for the Negotiations on the Revision of the Gothenburg Protocol under the Convention on Long-range Transboundary Air Pollution. Background paper for the 48th Session of the Working Group on Strategies and Review, Geneva, April 11-14, 2011

Cummings, R.G. and G.W. Harrison (1995). The measurement and decomposition of nonuse values: a critical review. Environmental and Resource Economics 5, 225-47.

Daily, G.C., S. Polasky, J. Goldstein, P.M Kareiva, H.A Mooney, L. Pejchar, T.H Ricketts, J. Salzman, and R. Shallenberger. 2009. Ecosystem services in decision making: time to deliver. Frontiers in Ecology and the Environment 7: 21—28.

De Smet, L., K. Devoldere and S. Vermoote, 2007. Valuation of air pollution ecosystem damage, acid rain, ozone, nitrogen and biodiversity. ENV.C.5/SER/2006/0114r. European Commission, DG ENV. 06/11867/SV. Arcadis Ecolas.

EPA, 1999. Costs and Benefits of the Clean Air Act.

Grimm, N.B., D. Foster, P. Groffman, J.M Grove, C.S Hopkinson et al., 2008. The changing landscape: ecosystem responses to urbanization and pollution across climatic and societal gradients. Frontiers in Ecology and Environment 2008; 6: 264–272.

Hanemann, W.M., 1995. Contingent valuation and economics. In: K. Willis and J. Corkindale (eds.). Environmental valuation: new perspectives. CAB International, Wallingford, UK. pp. 79-117.

Hettelingh, J.P., M. Posch, J. Slootweg, G. J. Reinds, T. Spranger and L. Tarrason, 2007. Critical Loads and Dynamic Modelling to Assess European Areas at Risk of Acidification and Eutrophication. Water, Air & Soil Pollution 7:379–384.

Holland, M., G. Mills, F. Hayes, A. Buse, L. Emberson et al., 2002. Economic Assessment of Crop Yield Losses from Ozone Exposure. Centre for Ecology and Hydrology, under contract EPG 1/3/170, UNECE International Cooperative Programme on Vegetation.

Little, I.M.D. and M.F. Scott, 1976. Using shadow prices. Heinemann, London.

MA, 2003. Millennium Ecosystem Assessment. Ecosystems and human well-being: a framework for assessment. Report of the conceptual framework working group of the Millennium Ecosystem Assessment. Island Press, Washington, DC.

MA, 2005. Millennium Ecosystem Assessment. Ecosystems and human well-being: Global Assessment Report Vol. 1 to 3. Island Press, Washington, DC

Munasinghe, M. and A. Schwab, 1993. Environmental economics and natural resource management in developing countries. World Bank, Washington, DC.

Pearce, D.W., and R.K. Turner, 1990. Economics of natural resources and the environment. Harvester Wheatsheaf, Herfordshire, UK.

Pearce, D.W. and A. Howarth, 2000. Technical report on methodology: cost benefit analysis and policy responses. RIVM report 481505020

Scheffer, M, S. Carpenter, J.A. Foley, C. Folke, and B. Walker, 2001. Catastrophic shifts in ecosystems. Nature 413: 591-596

Scheffer, M., and S.R. Carpenter, 2003. Catastrophic regime shifts in ecosystems: linking theory to observation. Trends in Ecology and Evolution 18: 648-656.

Smart, J.C.R., K. Hicks, T. Morrissey, A, Heinemeyer, M.A. Sutton and M. Ashmore. Applying the ecosystem service concept to air quality management in the UK, a case study for ammonia. Environmetrics 2011, 22, 649-661.

TEEB, 2010. The Economics of Ecosystems and Biodiversity. Ecological and Economic Foundations. Edited by Pushpam Kumar. Earthscan, London and Washington DC.

Turner, R.K., C.J.M. van den Bergh, T. Soderqvist, A. Barendregt, J. van der Straaten, E. Maltby, E.C. van Ierland, 2000: Ecological-economic analysis of wetlands: scientific integration for management and policy. Ecological Economics 35: 7-23.

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