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Effects of long range transported air pollution (LRTAP) on freshwater ecosystem services

International Cooperative Programme on Assessment and Monitoring Effects of Air Pollution on Rivers and Lakes

Convention on Long-Range Transboundary Air Pollution

Norwegian Institute for Water Research
– an institute in the Environmental Research Alliance of Norway

REPORT

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Abstract
Acid deposition affects the services provided by freshwater ecosystems. The loss of sport fisheries is by far the most important ecosystem service affected. Both inland fisheries (such as brown trout) and anadromous fish (such as Atlantic salmon) are sensitive to acidification. Damage and loss of fish populations has large ramifications on other ecosystem services such as tourism, biodiversity, aesthetic value and cultural value. On-going work in both Europe and Canada indicates significant improvements in our ability to value in monetary terms the numerous potential benefits and costs of acid deposition abatement. There are still large gaps in the understanding of the nature and value of acid deposition impacts. Despite the fact that some economic-evaluation modelling capacity currently exists, economic-evaluation models for acid deposition do not adequately account for environmental benefits resulting from abatement.

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CONVENTION ON LONG-RANGE TRANSBOUNDARY AIR POLLUTION

INTERNATIONAL COOPERATIVE PROGRAMME ON ASSESSMENT AND MONITORING EFFECTS OF AIR POLLUTION ON RIVERS AND LAKES

Effects of long range transported air pollution (LRTAP) on freshwater ecosystem services

Prepared at the ICP Waters Programme Centre
Norwegian Institute for Water Research
Oslo, May 2013

Preface

The international cooperative programme on assessment and monitoring of air pollution on rivers and lakes (ICP Waters) was established under the Executive Body of the UNECE Convention on Long-range Transboundary Air Pollution (LRTAP) in July 1985. Since then ICP Waters has been an important contributor to document the effects of implementing the Protocols under the Convention. Numerous assessments, workshops, reports and publications covering the effects of long-range transported air pollution has been published over the years.

The ICP Waters Programme Centre is hosted by the Norwegian Institute for Water Research (NIVA), while the Norwegian Climate and Pollution Agency (Klif) leads the programme. The Programme Centre's work is supported financially by Klif.

The main aim of the ICP Waters Programme is to assess, on a regional basis, the degree and geographical extent of the impact of atmospheric pollution, in particular acidification, on surface waters. More than 20 countries in Europe and North America participate in the programme on a regular basis.

ICP Waters is based on existing surface water monitoring programmes in the participating countries, implemented by voluntary contributions. The ICP Waters site network is geographically extensive and includes long-term data series (more than 20 years) for many sites. The programme yearly conducts chemical and biological intercalibrations.

This report introduces the concept of ecosystem services, and describes the potential effects of acid deposition to services provided by freshwater ecosystems. Several examples are given from studies in Europe and North America. The report was prepared by Silje Holen, an environmental economist, Richard Wright, a biogeochemist, and Isabel Seifert, an environmental economist; all are researchers at NIVA. The work was supported by the ICP Waters Programme Centre.

Brit Lisa Skjelkvåle
ICP Waters Programme Centre

Oslo, May 2013

Contents

Summary

This report introduces the concept of ecosystem services, and describes the potential effects of acid deposition to services provided by freshwater ecosystems. The links between nature and the economy are often described using the concept of ecosystem services, or flows of value to human societies as a result of the state and quantity of natural capital. Ecosystem services fall into four broad categories: provisioning services, regulating services, supporting services and cultural services. Provisioning services are the products obtained from ecosystems, such as food, fibre and wood/fuel. Regulating services refer to the regulation of e.g. climate, water quantity, and water quality. Cultural services are the nonmaterial benefits people obtain from ecosystems through spiritual enrichment, cognitive development, reflection, recreation, and aesthetic experiences. Supporting services are those that are necessary for the production of all other ecosystem services.

The concept of ecosystem services has arisen in response to an increased need for making visible human dependency on nature and ecosystems, in order to ensure sustainable management and avoid irreversible damage to the ecosystems that ultimately will damage human welfare. Ecosystem services can capture a wider set of costs and benefits, not traditionally valued in economic analysis.

The value of natural resources is often considered within the framework of Total Economic Value (TEV), and this framework can be used also in the economic valuation of ecosystem services. TEV acknowledges that environmental resources have value beyond their direct consumption. Failure to consider all sources of value results in underestimates of the benefits of pollution abatement and inhibits sustainable development planning.

The loss of sport fisheries is by far the most important ecosystem service affected by acidification of freshwaters. Both inland fisheries (such as brown trout) and coastal anadromous fish (such as Atlantic salmon) are sensitive to acidification. Damage and loss of fish populations has large ramifications on other ecosystem services such as tourism, biodiversity, aesthetic value and cultural value.

On-going work in both Europe and Canada indicates significant improvements in our ability to value in monetary terms the numerous potential benefits and costs of acid deposition abatement. Improvements in modelling and valuation increase understanding of the socioeconomic impacts of acid deposition policies. This can support more effective and efficient management strategies. Various economic valuation methods are evaluated in terms of their suitability for assessment of different freshwater ecosystem services impacted by acidification. The impacts of acidification are not instantaneous, and thus there is a need to aggregate the costs of the impacts over time. This is the present value of the impacts over time, discounted at the social discount rate. How should today’s costs and benefits be weighted towards future costs and benefits? The discount rate adjusts downward costs and benefits occurring in the distant future.

While effects on aquatic and terrestrial ecosystems due to acidification have been widely studied for decades, they have seldom been evaluated in economic terms. Valuation research in the context of air pollution and freshwaters has focused largely on damage to fish populations, particularly popular angling species such as trout and salmon. Examples of such studies come from several countries affected by acidification of freshwaters. An overview of these studies is given here.

The Tovdal River, Norway, exemplifies a case where long-range transboundary air pollution has caused severe damage to freshwater ecosystems, and the ecosystem services. In particular is the loss of the native salmon population in the river as well as brown trout populations from many of the 300 lakes in the river basin.

Although environmental economic research has come a long way, there are still large gaps in the understanding of the nature and value of acid deposition impacts. Despite the fact that some economic-evaluation modelling capacity currently exists, economic-evaluation models for acid deposition do not adequately account for environmental benefits resulting from abatement. It appears that there has never been a thorough analysis of the total costs and benefits of reducing acid deposition in terms of ecosystem services.

1. Introduction

The overall aim of this report is to assess current prospects for evaluation of damage to freshwater ecosystem services by long range transported air pollution, in particular acid deposition (sulphur and nitrogen).

We first summarize the impacts of LRTAP on freshwater ecosystem services. We then set up a framework for assessment and economic valuation of impacts of long range transported air pollution (LRTAP) on freshwater ecosystem services.

The case of the Norwegian river Tovdalselva will be used to illustrate how the assessment of ecosystem services can be approached at a local level, based on existing data and information.

We give examples from the literature on ecosystem services valuation in the context of long-range air pollution in Europe and North America, and describe the main challenges of economic evaluation of damage to freshwater ecosystems. We present suggestions for future research needs in terms of economic valuation and scientific understanding.

1.1 Acid deposition and acidification of freshwaters

Acid deposition has caused widespread acidification of freshwaters with damage to fish populations and other aquatic organisms. Acid deposition is caused by emissions of sulphur (S) and nitrogen (N) to the air, with subsequent dispersion and long-range transport in the atmosphere. During the past 200 years large areas of Europe and eastern North America have been affected by acid deposition and acidification of surface waters. More recently acid deposition has also become a problem in other parts of the world, such as eastern China, Japan and other highly-industrialised regions.

Acidification of freshwaters requires two factors: acid deposition and acid-sensitive catchment ecosystems. Because most precipitation falls on the terrestrial parts of the catchment, lake and streamwater chemistry is strongly affected by soil properties. Acid sensitive waters are typically located in catchments with highly siliceous soils with low acid neutralising capacity. In Europe, water acidification has been most widespread in Fenno-scandia, where thousands of freshwaters have been affected (Skjelkvåle et al., 2006). Upland areas elsewhere in Europe such as the UK, central and eastern Europe and the Alps have also been affected. In North America, widespread acidification occurs in southeastern Canada (Ontario, Quebec, and the Atlantic provinces) (Jeffries, 1997), and eastern United States (upland areas of New York, the New England states and the southern Appalachian Mountains) (Driscoll et al., 2001).

Acidification causes major changes in aquatic ecosystems. Most prominent is the damage and loss of fish populations. In Norway, acidic deposition caused the loss of brown trout populations in thousands of lakes and extinction of native salmon populations from seven major rivers (Hesthagen et al., 1999; Jensen and Snekvik, 1972). A survey conducted in the 1990s in Fenno-Scandia showed that acidification has affected fish populations in more than 10 000 lakes (Tammi et al., 2003). Acidification affects all trophic levels in aquatic ecosystems, including benthic invertebrates, planktonic fauna, planktonic and attached algae. Acid sensitive species decline in abundance and disappear, while acid tolerant species increase in abundance and invade. Adverse effects on aquatic ecosystems have been reported from Norway, Sweden, Finland, the UK, France, Germany, the Czech Republic, Slovakia, Poland, Switzerland and Italy in Europe, and in parts of Canada and the United States in North America.

In the 1970s the widespread acidification of freshwaters and damage to fish populations in Europe prompted calls for reductions in emissions of air pollutants. In 1979 the United Nations Economic Council for Europe (UNECE) founded the Convention on Long-range Transboundary Air Pollution (LRTAP) (www.unece.org). The Convention strives to reduce air pollution problems through scientific collaboration and policy negotiation. Protocols implemented since the 1980s have reduced the emissions of sulphur and nitrogen to the atmosphere in Europe by about 80% and 50%, respectively, since the peak years in the 1970s and 1980s (Schöpp et al., 2003) (***Figure 1***).

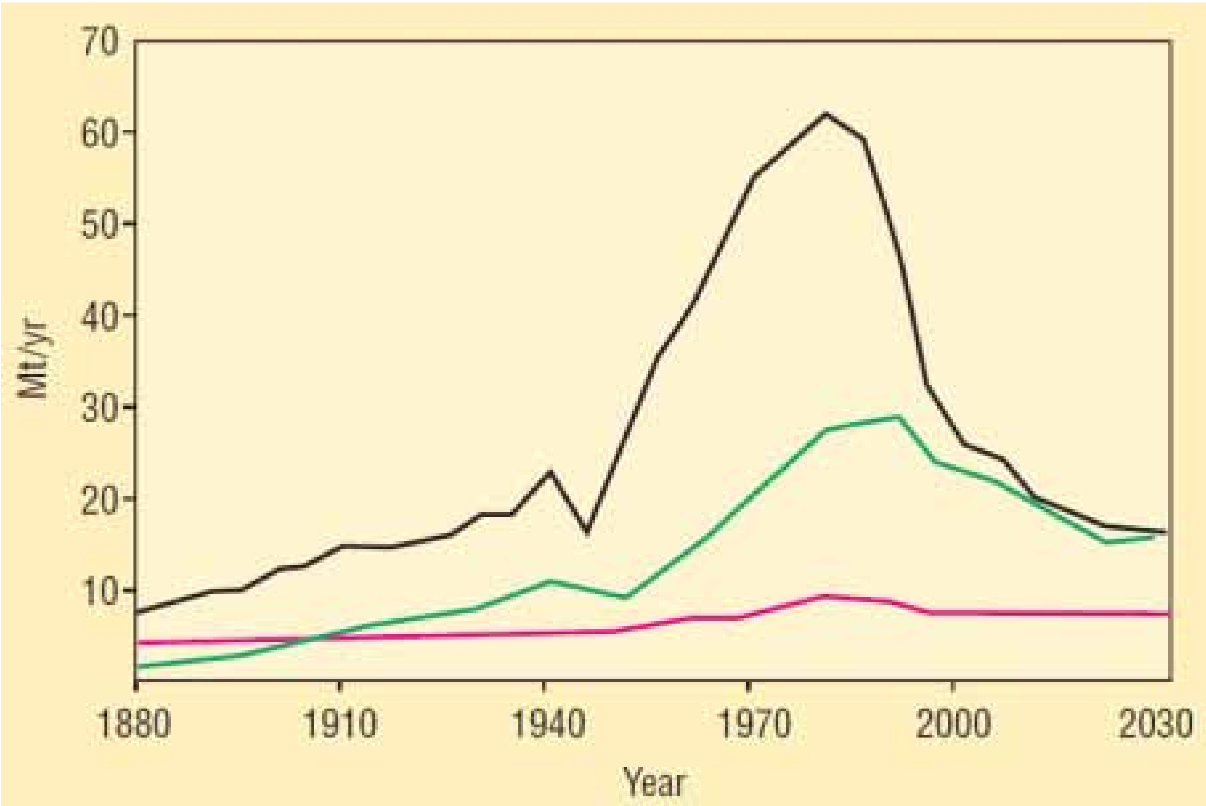


Figure 1. *Emissions of SO_2 (black line), NOx (green line) and NHy (red line) in Europe over the period 1880-2030 (from Schöpp et al. (2003). The prognosis for future years is based on implementation of the 1999 Gothenburg protocol to the LRTAP convention. The recent amendment signed in 2012 entails additional reductions in the future of both S and N emissions.*

Emissions of air pollutants have decreased also in eastern North America (***Figure 2***). In the United States the 1990 Clean Air Act Amendments targeted control of acid deposition, and resulted in decreased S deposition at acid-impacted regions in the eastern US (Kahl et al., 2004). In Canada agreements to reduce emissions of S were established in the late 1980s and in 1991 the Canada-US Air

Quality Agreement was signed, to limit transboundary transport of air pollutants. These have resulted in reduced S deposition in acid-impacted areas of eastern Canada. In Canada, SO_2 emissions have decreased by about 50% and in the U.S. by about 40% over the period 1980 to 2000 (Morrison and Carou, 2004). NOx emissions have decreased only slightly over this same period.

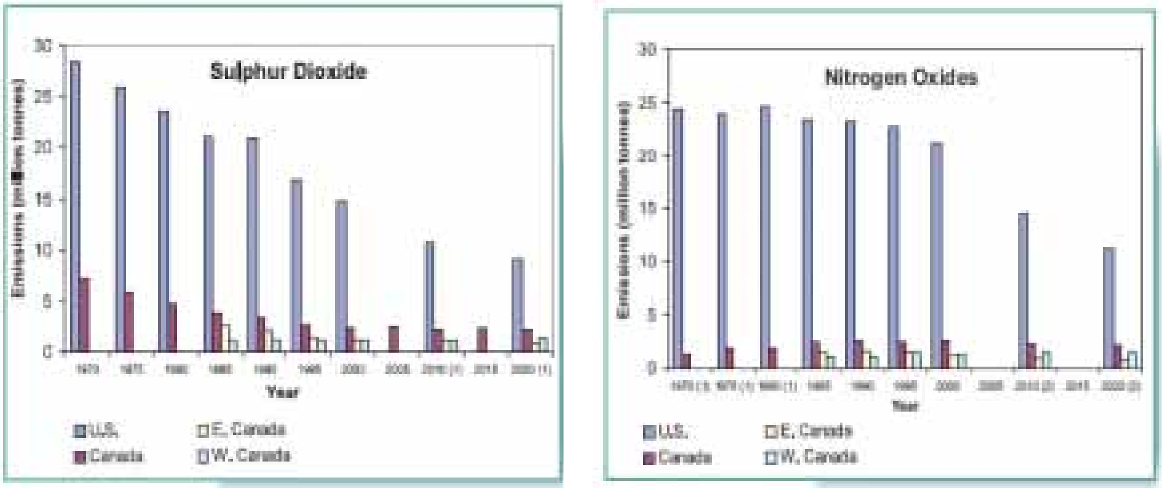


Figure 2. *Emissions of SO2 and NOx in the U.S. and Canada. Source: Morrison and Carou (2004).*

Freshwaters in Europe and North America have begun recovery in response to the reduced acid deposition. Chemical conditions have improved since the mid-1980s in acidified lakes and streams. These changes have been extensively documented by ICP-Waters and national monitoring programmes (Evans and Monteith, 2001; Hruška et al., 2002; Jeffries et al., 2003; Kernan et al., 2010; Skjelkvåle et al., 2007; Skjelkvåle et al., 2003; Skjelkvåle et al., 2005; Skjelkvåle et al., 1998; Stoddard et al., 1999; Wright et al., 2005). In many of these waters the biota also shows signs of recovery, but often lagging many years behind the chemical recovery. Examples include the Killarney lakes near Sudbury in southern Ontario Canada (Gunn and Sandøy, 2003), a salmon river in Norway (Kroglund et al., 2001), and acidified waters in the UK (Monteith et al., 2005).

Text box 1. *The rise and fall of acidification at Lake Lille Hovvatn, southernmost Norway. This acidified lake is typical of thousands of acid-sensitive freshwaters in Norway. The native brown trout population was lost in the 1940s due to chronic acid deposition. Reduced sulphur deposition since the peak years in the 1970s has resulted in gradual chemical recovery, but conditions still are insufficient to support reproduction of brown trout. Shown are the concentrations of sulphate (SO_4), ANC (acid neutralising capacity), Al^{n+} (inorganic aluminium), and the pH measured (1974-2011) and simulated (1870-2050) by the MAGIC model. Modified from: Hindar and Wright (2005).*

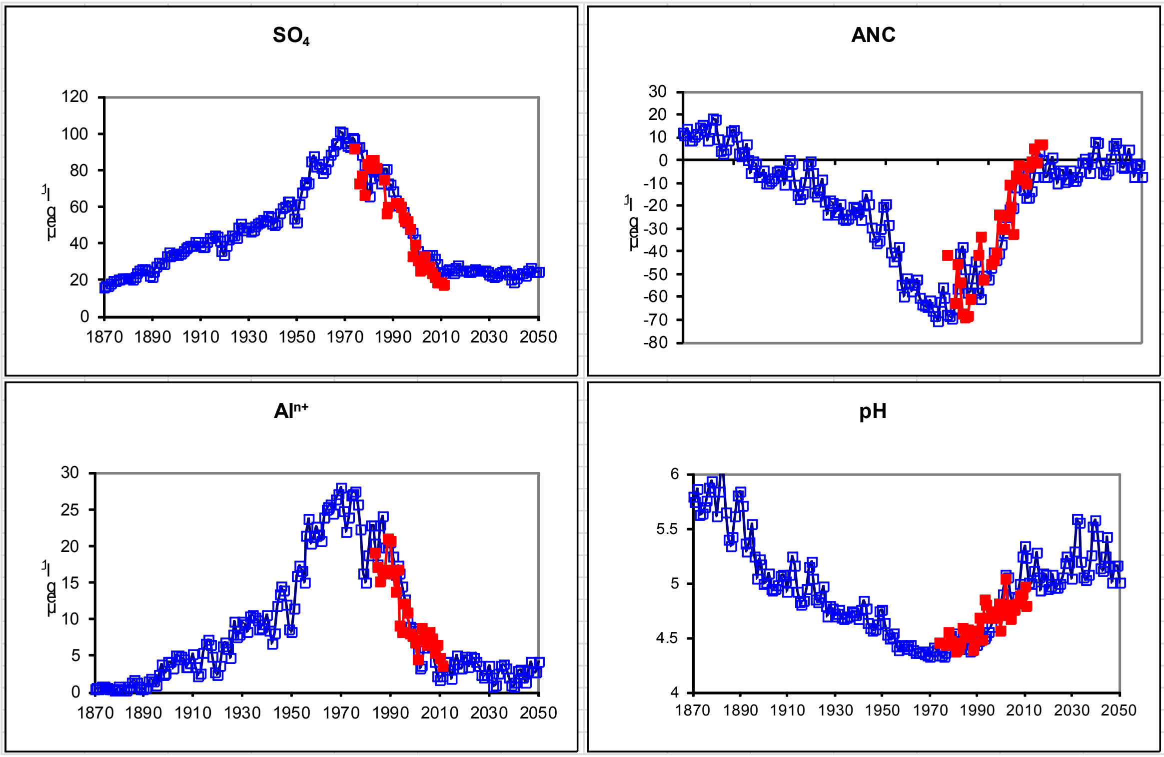


Photo: Richard Wright

Text box 2. *Chemical and biological recovery at Lake Saudlandsvatn, southern Norway. Shown are long-term trends in deposition of sulphate, acid neutralising capacity and pH of lakewater, and the responses three acid sensitive species – brown trout (catch per unit effort Cpue), the mayfly Baetis rhodani, and the zooplankter Daphnia longispina (from Hesthagen et al. (2011).*

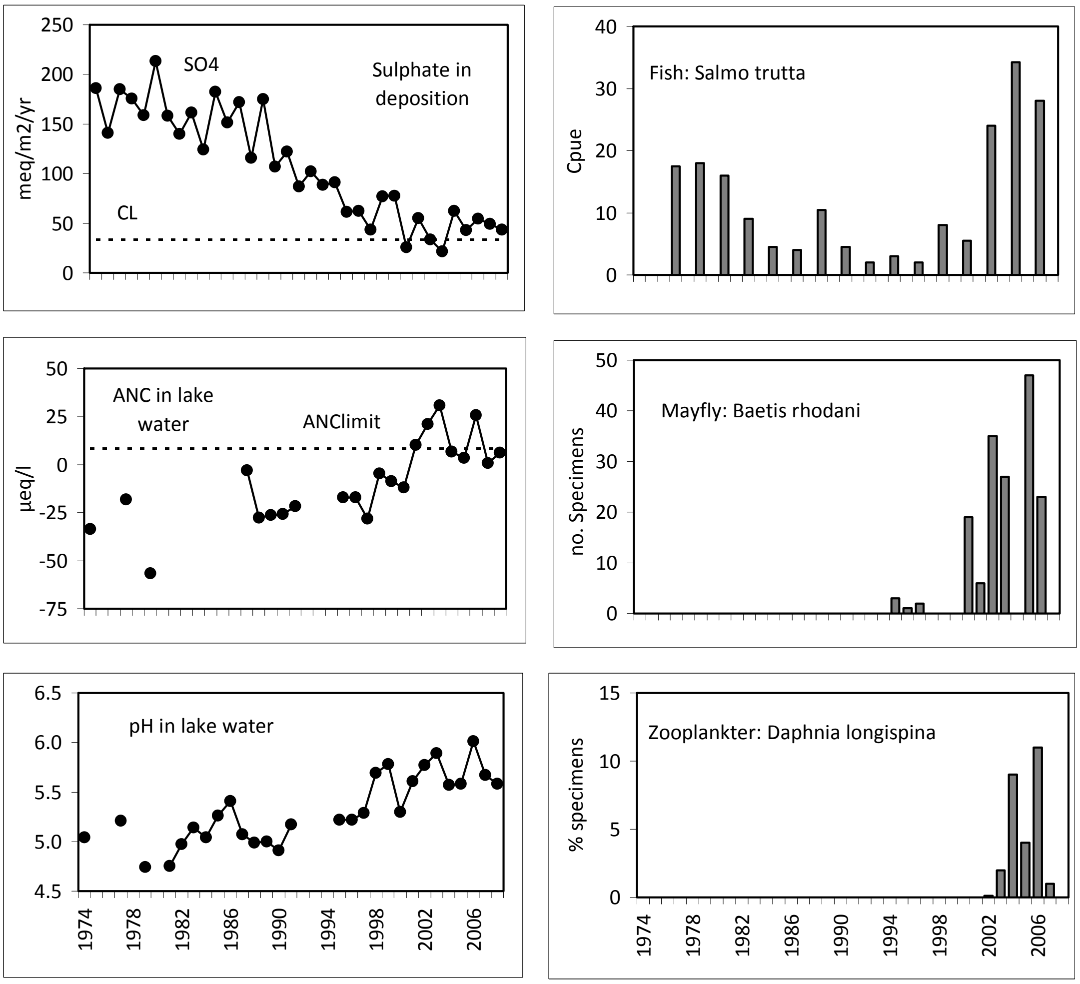


Photo: Arne Fjellheim

1.2 The concept of ecosystem services

The links between nature and the economy are often described using the concept of ecosystem services, or flows of value to human societies as a result of the state and quantity of natural capital (MEA, 2005; TEEB, 2010). The Millennium Ecosystem Assessment (MEA, 2005) conducted under the auspices of the United Nations had as its objective to assess the consequences of ecosystem change for human well-being and the scientific basis for action needed to enhance the conservation and sustainable use of those systems and their contributions to human well-being.

The Millennium Ecosystem Assessment grouped ecosystem services into four broad categories: provisioning services, regulating services, supporting services and cultural services (MEA, 2005) (***Figure 3***). Provisioning services are the products obtained from ecosystems, such as food, fibre and wood/fuel, and other products such as the provision of fresh water. Regulating services refer to the regulation of e.g. climate, water quantity (ground water recharge, occurrence of floods etc.), water quality and diseases and are related to the impact of ecosystems on greenhouse gas exchange and buffering and filtering capacity of the soil affecting water and element fluxes. Supporting services are those that are necessary for the production of all other ecosystem services. They differ from provisioning, regulating, and cultural services in that their impacts on people are either indirect or occur over a very long time, whereas changes in the other categories have relatively direct and short-term impacts on people. Some services, such as erosion control, can be categorized as both a supporting and a regulating service, depending on the time scale and immediacy of their impact on people. For example, humans do not directly use soil formation services, although changes in soil indirectly affect people through the impact on the provisioning service of food production. Similarly, climate regulation is categorized as a regulating service since ecosystem changes can have an impact on local or global climate over time scales relevant to human decision-making (decades or centuries), whereas the production of oxygen gas (through photosynthesis) is categorized as a supporting service since any impacts on the concentration of oxygen in the atmosphere would only occur over an extremely long time. Some other examples of supporting services are primary production, production of atmospheric oxygen, soil formation and retention, nutrient cycling, water cycling, and provisioning of habitat. Cultural services are the nonmaterial benefits people obtain from ecosystems through spiritual enrichment, cognitive development, reflection, recreation, and aesthetic experiences.

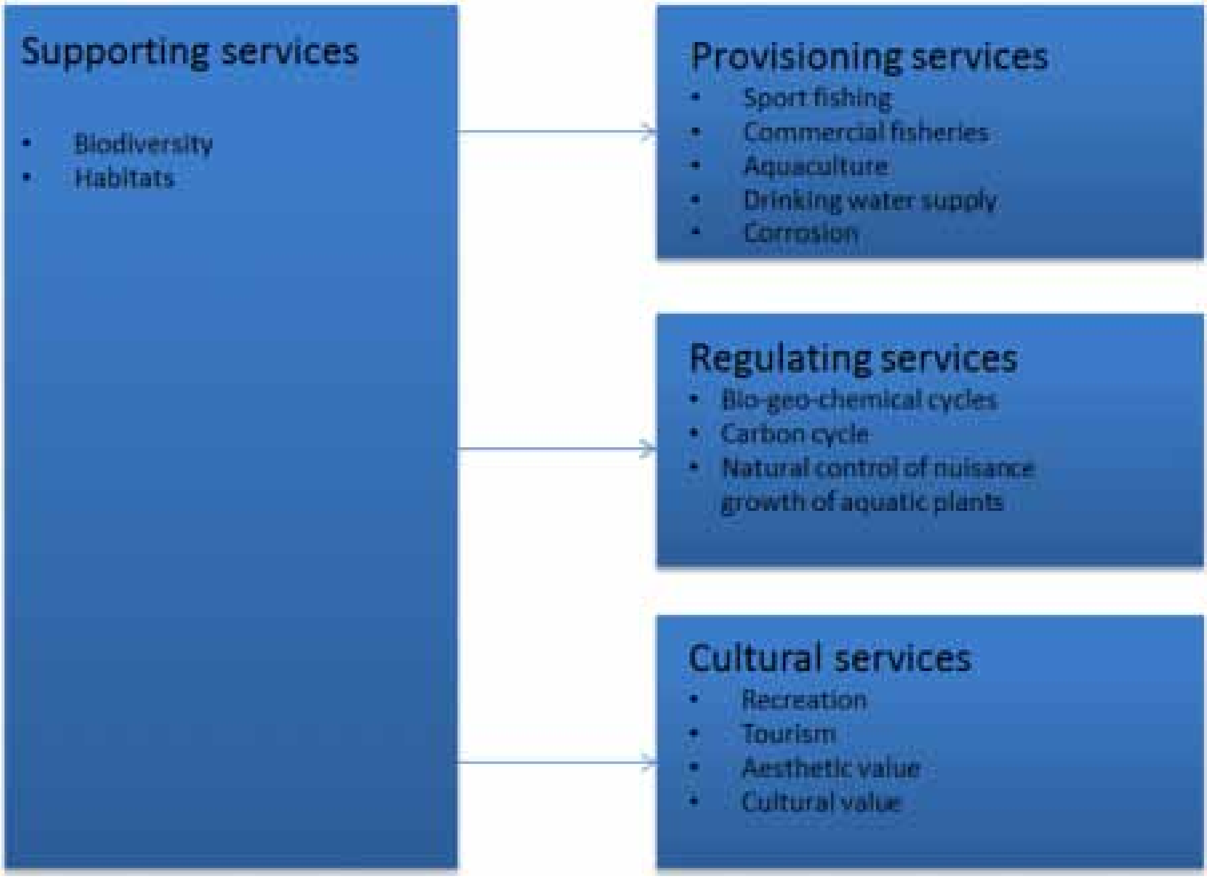


Figure 3. *Freshwater ecosystem goods and services at risk from acidification.*

The quantity and quality of the delivered ecosystem services depend on the status of the ecosystems, future exposure to pressures, the adaptation capacity of these ecosystems and the willingness and resources of the society to maintain the ecosystems and their services. The status of ecosystems is determined by biogeochemical characteristics, which reflect also the current status of pressures affecting the system. The status can be described by various biological quality elements and chemical parameters, as the diversity and density of organisms populating the ecosystem and chemical water, soil and air quality parameters.

1.3 Why valuation of ecosystem services?

The low visibility of biodiversity values has often encouraged inefficient use or even destruction of the natural capital that is the foundation of our economies (TEEB, 2010). The concept of ecosystem services has arisen in response to an increased need for making visible human dependency on nature and ecosystems, in order to ensure sustainable management and avoid irreversible damage to the ecosystems that ultimately will damage human welfare. There is not any straightforward link between the concept of ecosystem services and consequences of ecosystem change for human well-being and the monetary valuation of these changes. However, there is an increasing focus on the relationship between ecosystem services and their economic value and especially the intrinsic value of nature. While the Millennium Ecosystem Assessment (MEA, 2005) identifies ecosystem services from a human point of view with a focus on how the ecosystems contribute to human welfare, The Economics of Ecosystems and Biodiversity (TEEB, 2010) is a global initiative focused on drawing attention to the economic benefits of biodiversity. The TEEB study was launched by Germany and the European Commission in response to a proposal by the G8+5 Environment Ministers in 2007, to develop a global study on the economics of biodiversity loss. The study highlights the cost of biodiversity loss and ecosystem degradation and brings together expertise from ecology, economics and development to support the mainstreaming of biodiversity and ecosystem considerations into policy making. The TEEB approach is about recognizing, demonstrating and capturing value (***Figure 4***).

The Economics of Ecosystems and Biodiversity (TEEB)		
Recognizing value	**Demonstrating value**	**Capturing value**
Recognizing value in ecosystems, landscapes, species and other aspects of biodiversity is a feature of all human societies and communities, and is sometimes sufficient to ensure conservation and sustainable use. This may be the case especially where the spiritual or cultural values of nature are strong.	Demonstrating value in qualitative and quantitative terms is, nevertheless, often useful for policymakers and others, such as businesses, in reaching decisions that consider the full costs and benefits of a proposed use of an ecosystem, rather than just those costs or values that enter markets in the form of private goods.	Capturing value involves the introduction of mechanisms that incorporate the values of ecosystems into decision making, through policy incentives and price signals.

Figure 4. *The TEEB approach: recognizing, demonstrating and capturing value (modified from TEEB (2010)).*

Many of the freshwater ecosystem services generated by natural capital are externalities and are thus not given a price in markets. Often, society would benefit from greater protection of ecosystems and their services. Public policy has a crucial role to play in regulating or influencing markets so as to prevent them from producing unfortunate societal outcomes. Perhaps the most important basis for supporting a policy that would protect otherwise threatened ecosystem services, is evidence that society gains more value from such protections than it gives up. Providing such evidence requires an understanding of the biophysical processes involved, that is, the various services offered by the freshwater ecosystem in question as well as an assessment of the values to society of these ecosystem services.

Socio-economic benefit and cost analyses can support the decision-making process by assessing the impacts of policy implementation in common terms. Economic approaches and techniques, as well as the use of a common monetary scale can allow for a more thorough appreciation of the extent of acid deposition impacts and provide policy makers with a way to objectively compare the costs and benefits of various abatement options (Bourassa et al., 2004).

1.4 Ecosystem services and acidification

Estimation of economic impacts and the costs and benefits of controlling acid rain has been an issue in Europe since the the 1970s. However, uncertainties in the relationship between deposition and effects were so large that the role of cost-benefit analysis has been limited (Menz and Seip, 2004). Ecosystem services can capture a wider set of costs and benefits, not traditionally valued in economic analysis (TEEB, 2010). Economic evaluation of damage to freshwater ecosystems by long-range transported air pollution is not straightforward because many of the ecosystem services are not traded in markets. There are many limitations of the economic evaluation of damage to ecosystems. Increased scientific knowledge improves our capacity to value ecosystem damage or recovery, but some fundamental difficulties remain. Economic evaluation is limited to instances where the impacts of ecosystem damage on human welfare can be recognized and measured scientifically. Some aspects of acidification and recovery may also be so complex and/or uncertain that valuation scenarios may be too simplistic or even misleading.

On-going work in both Europe and Canada indicates significant improvements in our ability to value in monetary terms the numerous potential benefits and costs of acid deposition abatement. Improvements in modelling and valuation increase understanding of the socioeconomic impacts of acid deposition policies. This can support more effective and efficient management strategies (Bourassa et al., 2004). Bourassa et al. (2004) present a generic approach to environmental valuation for acid rain. This approach follows four steps: pollution is emitted, pollution changes ambient air quality, ambient air quality has physical effects on humans and the environment, and physical effects are assigned monetary value based on their links to human beings (***Figure 5***).

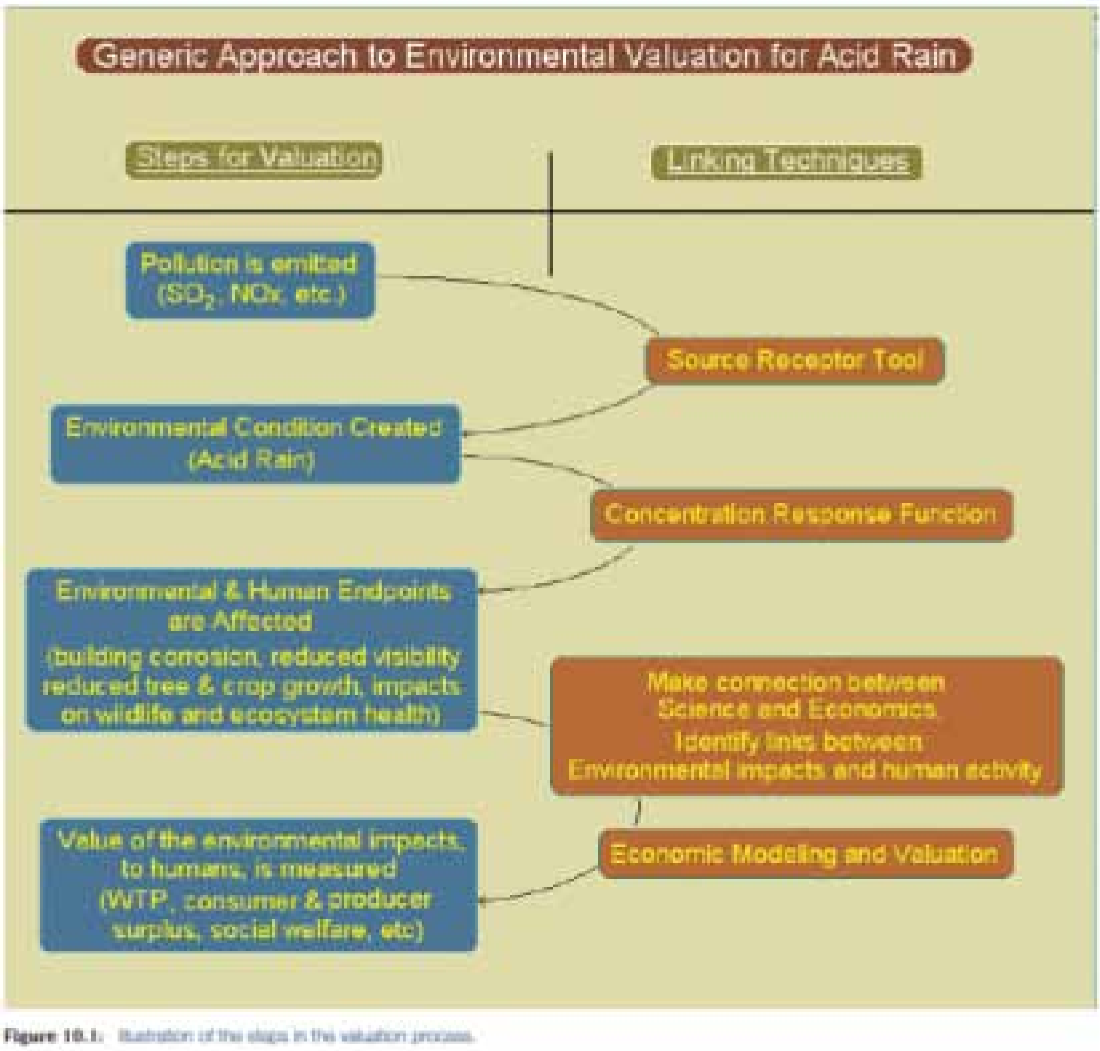


Figure 5. *Illustration of the steps in the valuation process for acid rain (from Bourassa et al. (2004)).*

2. Freshwater ecosystem services at risk from acidification

The loss of sport fisheries is by far the most important ecosystem service affected by acidification of freshwaters. Both inland fisheries (such as brown trout) and coastal anadromous fish (such as Atlantic salmon) are sensitive to acidification. Damage and loss of fish populations has large ramifications on other ecosystem services such as tourism, biodiversity, aesthetic value and cultural value.

2.1 Provisioning services

Sport fishing is impaired in regions with acidified waters. Also commercial fishing is affected. Acid water is disadvantageous to the aquaculture industry because smolt production requires abundant freshwater of suitable quality for reproduction of trout and salmon. Water acidification also increases corrosivity, and thus increases corrosion of turbines for hydropower production and adversely affects other industrial uses of water.

2.2 Regulating services

Biogeochemical cycles

Lakes and rivers are important parts of the biogeochemical cycles of several major elements such as carbon (C), nitrogen (N) and sulphur (S). N is removed from the water by such processes as denitrification, uptake by algae and other plants, and in part retained in the sediment. Removal of N thus means that less N flows to the sea, where it is often a pollutant and the growth-limiting nutrient. S is also removed from water in lakes and rivers, mostly by reduction at the sediment-water interface and storage in the sediments. Removal of N and S is related to the water flushing time (Kaste and Dillon, 2003). Acid deposition increases the flux of SO_4 and NO_3 from terrestrial catchments; a fraction of both is lost in lakes and rivers.

Lakes and rivers also modify the dissolved organic carbon (DOC) concentrations. DOC flowing from soils and wetlands in the terrestrial catchment is broken down by sunlight and microbial activity in lakes and rivers (Gennings et al., 2001; Wu et al., 2005). Acid deposition is thought to depress the concentrations of DOC in runoff from terrestrial catchments (Monteith et al., 2007).

Carbon cycle

Lakes sequester C in sediments, and thus act as a sink for atmospheric CO_2 (Cole et al., 2007).

Natural control of nuisance growth of aquatic plants

Nuisance growth of the aquatic macrophyte Juncus bulbosus has been observed in an increasing number of lakes and rivers in Europe. Among the consequences of such nuisance growth are reduced biodiversity, reduced suitability of the ecosystems for fish spawning, clogging of hydropower inlet screens and reduced suitability of the ecosystems for recreational use such as fishing, boating and bathing (Moe et al., 2013). For rivers an enhanced supply of N appears to be a trigger for enhanced growth (Schneider et al., 2013). Acid deposition may thereby promote growth of *Juncus bulbosus*.

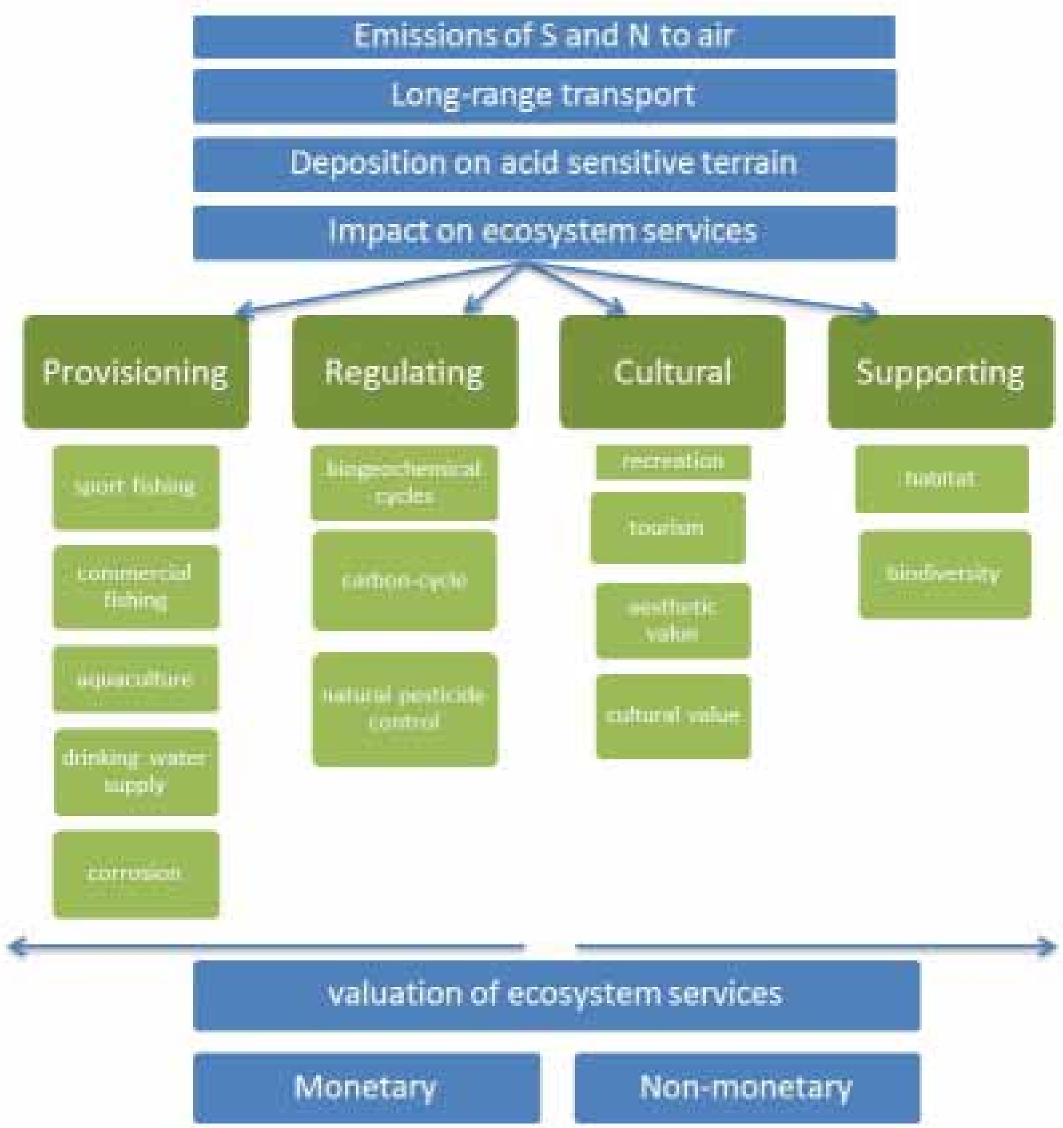
2.3 Cultural services

Acidification of freshwaters has major effects on use of the ecosystems for recreation. Tourism declines in acid impacted regions. Cultural losses included the abandonment of traditional gathering of fish from mountain lakes by the local farmers, as a supplement to their food resources and income. In Norway the disappearance of salmon in major rivers beginning in the 1920s meant the end of century-old traditions of salmon fishing and all the secondary supporting activities. The English lords no longer came to fish in rivers such as Tovdal River in southernmost Norway.

2.4 Supporting services

Of supporting services, impacts on biodiversity are probably the most significant effect of acidification. Acidification affects all trophic levels and all organism groups in freshwaters. The numbers of species decreases and in the case of extirpation of fish, whole trophic levels can be lost. Acidified freshwaters no longer provide a suitable habitat for the full biodiversity characteristic of un-impacted waters.

Text box 3. Schematic links between emissions of air pollutants and impacts on freshwater ecosystem services.



Emissions of sulphur and nitrogen compounds to the atmosphere, followed by long-range transport and deposition of acid to sensitive terrain, causes acidification of freshwaters and damage to aquatic organisms. The many services provided to society by these ecosystems are thus affected. The loss of ecosystem services can be evaluated in terms of money: damage to fish means reduced income from commercial or sport fisheries, or in non-monetary terms, such as the diminishment of aesthetic value: a lake that has lost its fish is less attractive than a lake that has intact fish populations.

The concept of ecosystem services is simply a means by which all these different types of environmental effects and their societal impacts can be evaluated together in a systematic manner.

3. Framework for economic valuation of impacts of acidification

The value of natural resources is often considered within the framework of Total Economic Value (TEV) (Pearce and Moran, 1994), and this framework can be used also in the economic valuation of ecosystem services. TEV acknowledges that environmental resources have value beyond their direct consumption. Failure to consider all sources of value results in underestimates of the benefits of pollution abatement and inhibits sustainable development planning. The TEEB study uses the Total Economic Value (TEV) setup (Pearce and Moran, 1994) to systematize the valuation of economic benefits. Bateman et al. (2003) add the concept of bequest value. This modifies the value of an environmental good to include the value to those alive now of leaving the good for future generations. This then shows up as both a use value, and as a non-use value, on the basis that the future generations will get both kinds of use from the asset. The diagram below shows the various components of environmental value (***Figure 6***).

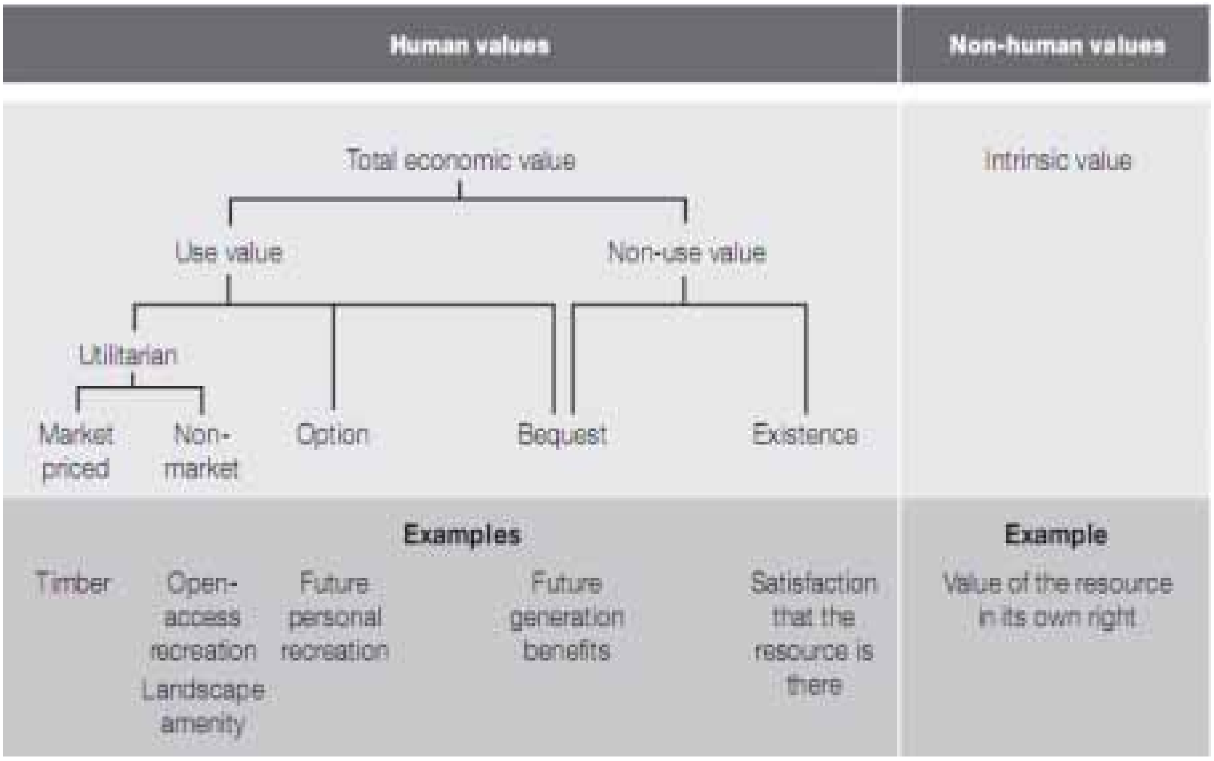


Figure 6. *Various components of environmental value (from Bateman et al. (2003)).*

Use Values:

- direct use value attributed to direct utilization of ecosystem services;
- indirect use value attributed to indirect utilization of ecosystem services, through the positive externalities that ecosystems provide;
- option value attributed to preserving the option to utilize ecosystem services in the future;
- altruistic value based on the welfare the ecosystem may give other people
- bequest value based on the welfare the ecosystem may give future generations

Non-use values:

- existence value attributed to the pure existence of an ecosystem
- bequest value based on the welfare the ecosystem may give future generations

The sum of these categories equals the TEV. However, these are the "economic" values done in an anthropocentric calculation. There is a category of non-economic values as well, often called intrinsic values. These values do not depend on human willingness to pay for them, but are intrinsic to the animal, ecosystem or other part of nature. Although TEV is a useful tool for illustrating the many sources of value for ecosystem services, it is not used directly in cost-benefit analysis (CBA) calculations. For CBA, change in total value has to be measured. For instance, to compute the benefits of reducing acid deposition, policy makers need to understand the change in environmental conditions resulting from the reduction in acid deposition and the value of that change. Thus, it is not TEV per se that is of use for policy oriented CBA; it is changes in TEV (Bourassa et al., 2004). Based on Armstrong et al. (2008), Anon. (2005), Toman (1998), Barton et al. (2012), MEA (2005) and TEEB (2010), the different economic valuation methods are evaluated in terms of their suitability for assessment of different freshwater ecosystem services impacted by acidification. ***Table 1*** provides an overview of which economic valuation methods can be used, and the data needed for the different valuation methods.

Table 1. *Overview of several economic valuation methods and the data requried for each.*

Valuation method	*Approach*	*Ecosystem services*	*Data needs*
Stated Preference (SP) methods	Willingness-to-pay (WTP) (or Willingness-to- accept (WTA) compensation) for changes in provision of ecosystem services are "stated" by respondents in surveys using structured questionnaires.	Supporting services Cultural services	Survey with scenario description and questions about WTP/WTA for specific services
Revealed Preference (RP) methods	Values are "revealed" through studying consumers' choices and the resulting price changes in actual markets that can then be associated with changes in provision of ecosystem services. These methods include the Hedonic Price method and The Travel Cost method.	Cultural services	Data of environmental quality attributes, consumers' choices and the resulting price changes in actual markets.
Production function approach	Ecosystems' marginal contribution to the final service is valued. When a change of ecosystem characteristics leads to off-site or downstream loss of services, biophysical damage functions of this "pressure-state-impact" relationship are used.	Provisioning services Regulating services Supporting services	Production function approach can be used when it is known how the ecosystem services contribute to the production of market goods.
Cost-based methods	Cost-based methods involve observing the real costs (including opportunity, prevention, avoidance, replacement, mitigation costs and damage costs) incurred by stakeholders in adapting to changes in ecosystems. These methods assume that the expenditures involved in preventing/mitigating or replacing losses of ecosystem services represents what people are willing to pay for the ecosystem services. However, WTP could be lower or higher than these costs.	Provisioning services Regulating services	Data on expenditures involved in preventing, mitigating or replacing lost ecosystem services.
Deliberative valuation methods	Consideration of values includes both monetary and non-monetary expressions. Deliberative methods can also systematically document non-monetary values. Multi-criteria analysis is one of these methods that have been used both as a qualitative deliberative method and as a method for indirect monetary valuation. Participatory, inclusive and deliberative fora such as focus groups provide a direct way of avoiding monetary valuation.	All services	No specific requirements
Benefit transfer (Value Transfer)	Refers to the use of secondary, existing study estimates, from any of the valuation methods mentioned above.	All services	Suitable, relevant and high-quality original valuation studies.

3.1 Discounting

How should today's costs and benefits be weighted towards future costs and benefits? This is a complicated and philosophical question, and the answer is far from obvious. In practice, a discount rate is used that discounts or adjusts downward costs and benefits occurring in the distant future in a cost-benefit analysis.
The impacts of acidification are not instantaneous, and thus there is a need to aggregate the costs of the impacts over time. This is the present value of the impacts over time, discounted at the social discount rate d. The social discount rate is the economic measure of how timing affects values (Birol et al., 2010). Large impacts will often occur far into the future and are usually assumed to be less likely to occur than smaller impacts. This is why, for example, natural disasters must be discounted in both time and probability.
Standard models for economic growth often assume a constant and positive social discount rate (Groom et al., 2005). Though a constant discount rate is in many ways an attractive trait, it also results in present and future generations being valued very differently. A positive constant discount rate takes less account of future generations' welfare than present generations' welfare and causes an exponentially decreasing emphasis on the future. By using a positive discount rate, large losses happening in the future will be discounted to minor present values. This could be a problem in terms of the principle of sustainable development in today's policy making. One possible alternative could be to not use any discount rate at all (Li and Lofgren, 2000). This will, however, lead to the problem of today's generation having to forfeit consumption for the welfare of future, presumably more wealthy and technologically advanced generations. Furthermore, there are so many future generations, that saving today will not make much difference in the welfare for future generations in general. The world's low income countries will also be hard hit by the decline in income. There are thus a multitude of ethical aspects surrounding the use of a zero discount rate that many do not find acceptable (Groom et al., 2005).

Hyperbolic discounting could be a possible solution to this problem. Experimental work by both psychologists and economists on individual choice has revealed that individuals discount the future at a declining rate that follows a hyperbolic path (see e.g. Karp (2005) and Weitzman (2001)). This means that events in the distant future will be strongly emphasized. Hyperbolic discount models have in common that they provide the decision maker with the ability to distinguish between events in the near and distant future. This is very relevant for long-term environmental problems such as ocean acidification. Hyperbolic discounting is not included in the recommendations for Cost Benefit Analysis (CBA) in Norway and United Kingdom. In Norway, a 4% discount rate is recommended for projects with duration < 40 years, 3 % for impacts arising between 40 and 75 years, and 2% for longer-term impacts (NOU, 2012). British authorities recommend discount rates falling stepwise from 3.5% to 1%, and they operate with six different classes of project length (Cairns, 2006). The answers can be dramatically different using a 2 % discount rate instead of a 1% discount rate. With a high discount rate, the future will be of little value in the calculation. The spatial discounting literature states that values that relate to what economists call non-use values should have much lower discount rates than use values (Brown et al. 2002).

4. Examples of valuation studies of freshwater ecosystem services affected by LRTAP

While effects on aquatic and terrestrial ecosystems due to acidification have been widely studied for decades, they have seldom been evaluated in economic terms. Valuation research in the context of air pollution and freshwaters has focused largely on damage to fish populations, particularly popular angling species such as trout and salmon.

4.1 United States

Violette (1985) and Mullen and Menz (1985) estimated changes in consumer surplus attributable to reductions in catch rates and fishable acreage using the Travel Cost (TC) method. Mullen and Menz (1985) valued recreational fishing lost as a result of acid deposition in the Adirondack Mountains in the northeastern United States. Based on lost 'angler days', annual costs of air pollution were estimated to be approximately $1.6 million. Criticisms of this study include the simple damage function used and the failure to model demand for angling as a function of more than just fish catch. For example, Forster (1986) argues that many other factors influence the decision to fish such as habit, connections to the site, and particularly attributes which enhance relaxation such as tranquillity and landscape. Despite these limitations Crocker (1985) went on to use these results in a scaling-up exercise for the eastern US.

Morey and Shaw (1992) used a individual Travel Cost model which included a wider selection of trip variables to analyse the impact of marginal reductions in air pollution on the value of angling in New York State. A key finding was that only anglers who considered catching trout as central to their recreational experience had a positive willingness-to-pay (WTP) for enhanced catches arising from pollution abatement. Consequently the overall benefits of abatement were small, with a 25% increase in catch valued at around only $3 per angler. Using the link between air pollution abatement and fish population health generated by the MAGIC model for different species, the EPA report (1999) estimated the benefits associated with the implementation of the Clean Air Act Amendment (CAAA) to vary from $12 to $49 million depending on the pH threshold for fish survival assumed. The costs were measured in terms of lost fishing days which were valued using Travel Cost-derived estimates of the value of a fishing day. Apart from the problems normally associated with using this method, the main criticism of this approach is the simplistic binary damage response function (fish/no fish) used which did not allow for intermediate damage levels.

Menz and Driscoll (1983) estimated the damage caused to fish stocks on the basis of the cost of 'restoring' water quality by liming. Epp and Alani (1979) devised a Hedonic Price model to investigate how stream pH affected property values in rural Pennsylvania and found that a one unit increase in pH increased property value by $1439.

Water quality ladders have been a popular approach to estimating marginal changes in water quality as they describe changes in a number of attributes affected by pollution. For example, (Carson and Mitchell, 1993) assessed the benefits of the US Clean Water Act in 1993 using a ladder that included distinct improvements ('unsuitable for activities' to 'boatable' to 'fishable' to 'swimmable').

The environmental attributes affected by pollution have been more explicitly investigated using choice experiments. For example, Johnson et al. (1997) used the Contingent Rating approach to estimate WTP for reduced pollution from electricity generation in the US. Attributes included restrictions on fish consumption bans, human health impacts, employment, and damage to sugar maple.

4.2 Canada

In Canada, other approaches have been used. Hough and J.E. Hanna Associates (1982) used the Productivity Method to estimate costs of acidification to commercial fisheries. Based on a number of different studies, including Talhelm et al. (1987) in Ontario and Quebec and Englin et al. (1991) in the north eastern U.S., the Air Quality Valuation Model (AQVM-E) estimates that the value to fishermen of a 1% change in acid deposition levels ranges from $0.003 to $0.07 per fisherman per day, with an expected value of $0.03 (Chestnut et al., 1999). These numbers may seem small given that, in 1996, average expenditure on fishing for those who participated in fishing was $27 per day (or $40 per day for those whose primary purpose was fishing) (Federal-Provincial-Territorial-Task-Force-on-the-Importance-of-Nature-to-Canadians, 2000). The relationship between acid deposition levels and the willingness of fishermen to pay to avoid its impacts may be nonlinear, or non-continuous. This highlights the need for further work to clarify the relationship between fish habitat and availability and its value to fishermen.

The decline of fish in lakes and rivers of eastern Canada has significant impacts on the fishing industry, particularly for Atlantic salmon. The effects of declining fish populations also have significant effects for recreational fishermen who spent $ 1.9 billion on fishing in 1996. There is some indication that damages to the overall integrity of the environment and ecosystem could be amongst the most economically significant impacts of acid deposition. In 1996, Canadians spent over $ 12 billion on nature related activities. This $ 12 billion in expenditures likely represents only the tip of the iceberg on the full value Canadians place on the environment (Bourassa et al., 2004).

4.3 Norway

In Norway, four studies of willingness-to-pay for changes in freshwater fish stocks have been conducted using reduction in acid rain as the reason for the changes. Strand (1981) and subsequently Navrud (1989) estimated WTP to protect all fish stocks from acidification in Norway. The Navrud study produced estimates of WTP (based on an open-ended format) of about 50 € per year for emission levels that corresponded to different levels of fish population status. Information was provided about the level of damage and the number of lakes damaged for each emission scenario based on a simple dose-response relationship between water chemistry and three fish population status levels (extinct, reduced, and no damage). Although the fish population model had good predictive power for the extinct and no damage levels, the relationship between water chemistry and the reduced damage category was less reliable. The study was repeated by Navrud (2001b) using a dichotomous choice format; WTP estimates increased to about 100 € per year. Navrud (2001b) provide links between exogenous variable and endogenous variable in economic models by using the strong sustainability indicator of critical loads of S and N to describe the environmental change to be valued in a CV survey. Through cost–benefit analysis (CBA) in Norway, Navrud (1993a), Navrud (1993b) and Navrud (1993c) showed that liming was socioeconomically profitable in certain water bodies.

4.4 Sweden

In Sweden, Bengtsson and Bogelius (1995) used benefits transfer from the Navrud studies to make CBAs on nine sites in Sweden. Although ecosystem benefits were disregarded, results showed that surface water liming was socioeconomically profitable in most cases.

Johansson and Kristrom (1988) used CV to estimate WTP for a programme that would almost completely eliminate S emissions in Sweden. Average WTP per respondent was about 500 € per year. Health impacts were also included hence no separate WTP estimates for ecosystems were provided.

Besides emission control, liming of lakes, streams, and wetlands is currently used to ameliorate acidification in Sweden. An alternative strategy is forest soil liming to restore the acidified upland soils from which much acidified runoff originates. Bostedt et al. (2010) found that surface water liming always is more efficient than even the most optimistic forest liming scenario by carrying out cost–benefit analysis which compared these liming strategies with a special emphasis on the time perspective for expected benefits. Benefits transfer was used to estimate use values for sport fishing and non-use values in terms of existence values. Their results show that large-scale forest soil liming is not socioeconomically profitable, while lake liming is, if it is done efficiently - in other words, if only acidified surface waters are treated. Thus, they show that the beguiling logic of "solving" a serious environmental problem at its source (aluminium in acidified runoff derived from forest soils) rather than continuing to treat the symptoms (aluminium toxicity to fish) has been misleading. They also found it to be of critical importance to know when, and to what extent, forest soil liming will improve the acidification status of surface waters. The wide range of expert views on this, from no improvements after 50 years to complete remediation of all acidified waters after 30 years, reflects the uncertainties.

Given the lack of studies on the benefits of preserving ecosystems from acidification, Bostedt et al. (2010) used valuations derived from the preferences of policymakers as a substitute. The effects of acidification on terrestrial and aquatic ecosystems are valued as a benefit transfer derived from a political agreement on air pollution reduction programs in the EU [see Protocol of Gothenburg on the Convention on Long-range Transboundary Air Pollution (1999) and European directive 2001/81/EC on National Emission Ceilings for some air pollutants]. The willingness to pay (WTP) of European society for protecting 1 ha of ecosystem from acidification and eutrophication was estimated using the standard price approach (SPA) by Vermoote and de Nocker (2003).

Costs and benefits from liming activities occur over different time periods and are thus not directly comparable. To convert costs and benefits to present values,Bostedt et al. (2010) used the discount rate during the first time period (years 0–30) on 3.5% and 3% in the second period (years 31–50), corresponding to discount rates recommended by the British government for CBAs of public projects (Cairns, 2006).

Paulrud (2001) estimated recreational benefits connected with sport fishing in the Bohus region of south-western Sweden. This region is part of the heavily acidified area where forest liming is contemplated.

4.5 UK

Macmillan et al. (1996) estimated the total economic value placed on biodiversity recovery in the semi-natural uplands of Scotland from acidification using CV. Approximately 1000 households were sampled by mail-shot for WTP for six scenarios which offered, in a split sample, alternative future ecosystem recovery levels (following abatement) and damage levels (under the status quo). Time-scale and uncertainty regarding future recovery were also investigated, with ecosystem change depicted by 'species boxes' that pictorially represented changes in the relative abundance of biodiversity. Average household WTP, elicited using a dichotomous choice format, for abatement of acid rain was about 400 € per household depending on the scenario. WTP was significantly influenced by the level of future damage but not by future recovery level. A weakness of the research, with respect to policy appraisal, was that the link between future emissions reduction and biodiversity levels was not underpinned by a reliable scientific model but by the informed opinion of scientific experts.

ECOTEC (1993) used similar pictorial representations of species affected by acidification in a study of the non-market benefits of reduced SO_2 emissions for the UK. Using in-person interviews the sampling frame included 1606 non-users (general public) and 587 users (anglers). Annual household WTP for non-users (additional water rates) was 45 €. These values are much lower than the estimates by Macmillan et al. (1996) above, but this may be partly due to differences in methodology. The ECOTEC study excluded very high WTP amounts and used an open-ended payment format which typically produces lower estimates of WTP than the dichotomous choice format used by Macmillan et al. (1996) However, one cannot rule out the possibility that the Scottish population values recovery more highly than other parts of the UK – perhaps due to their closer proximity to damaged areas.

The UK rod and line (r&l) salmon fishery is a private resource traded in the market place, hence market data for individual beats allows the links between fish catch, fishery value, and water quality to be explored. MacMillan and Ferrier (1994) used the Hedonic Price method to predict the economic benefits of recovery in the r&l salmon fishery in Galloway area, Scotland. Three alternative deposition scenarios for SO_2 (constant 1988 levels; a 60% reduction from 1980 levels by 2003; and a 90% reduction from 1980 levels by 2008) were linked to changes in water chemistry, fish population status and fish catch over a 50-year time scale using MAGIC, a process-based catchment model for acidification. The impact of increased catch on salmon values were then linked to the value of the fishery using a Hedonic Price relationship for the UK salmon fishery developed by Radford et al. (1991). The results were consistent with the assumption of diminishing marginal returns. Under scenario 1 (status quo) the market value of the fishery was predicted to decline gradually from 16 € million in 1988 to 15 € million in 2033 in response to declining catch. Under both scenario 2 and 3 SO_2 emission reductions initiate a relatively modest recovery in market value: under scenario 2 the market value of the fishery rose to 18 € million after 50 years.

Milner and Varallo (1990) also used Radford's survey of salmon r&l fisheries to estimate the cost of acidification in Welsh fisheries. Damage estimates were in the region of 1.3 to 6.5 € million. Their analysis relied on a simple presence/absence relationship between water chemistry and fish catch verified by the results of an angler questionnaire and fish population survey. The study did not consider the timing and extent of recovery.

Bateman et al. (2005) value benefits, but not costs, of liming projects in Scotland. They introduce the contingent valuation method for monetary valuation of individuals' preferences regarding changes to environmental goods. Approaches to the validity testing of results from such studies are discussed. These focus upon whether findings conform with economic-theoretic expectations, in particular regarding whether valuations are sensitive to the size (or 'scope') of environmental change being considered, and whether they are invariant to alterations in study design which are irrelevant from the perspective of economic theory. Bateman et al. (2005) apply such tests to a large sample study of schemes to alter the acidity levels of remote mountain lakes. Results suggest that, when presented with environmental changes about which respondents are concerned, their values exhibit scope sensitivity and conform to theoretical expectations, and therefore could be used for formulating policy. However, when presented with changes which respondents feel are trivial, their values fail tests of theoretical consistency and are not scope sensitive, and therefore cannot be used within economic appraisals. Bateman et al. (2005) find that qualitative focus group analyses are good indicators of whether a given change is likely to be considered trivial or not and therefore whether scope sensitivity tests are likely to be satisfied.

4.6 Finland

Iivonen et al. (1995) carried out a questionnaire among 800 residents around Lake Alinenjärvi and found that even non-fishing residents around the limed lake were willing to finance the cost of liming, due to opportunities for recreation and swimming in the lake with only minor usage for fishing.

4.7 Elsewhere in Europe

The willingness to pay (WTP) of European society for protecting 1 ha of ecosystem from acidification and eutrophication was estimated using the standard price approach (SPA) by Vermoote and de Nocker (2003). The SPA is based on the notion of using abatement costs of emission reductions as a proxy for revealed WTP for improvements in ecosystem health. Abatement costs were based on the most cost-effective emission reduction programs. Policymakers are implicitly assumed to act rationally and to carefully balance abatement costs with the benefits of the emission reductions. To estimate the WTP to protect 1 ha year^{-1} of ecosystem from acidification and eutrophication, the numbers of hectare of ecosystems where critical loads were exceeded for acidifying and eutrophying deposition, and the total costs to reduce the emissions, have been calculated. The WTP ha^{-1} ranges between 63 and 350 € ha^{-1} year^{-1} (Vermoote and de Nocker, 2003), the variation depending on the selected emission reduction program. TheWTP that best reflected the EU was based on the Protocol of Gothenburg and the European directive 2001/81/EC (Vermoote and de Nocker, 2003), and equalled 100 € ha^{-1} year^{-1}. As this estimate involves measures to protect ecosystems from both acidification and eutrophication, it is an overestimate of the WTP to protect ecosystems from acidification. On the other hand, as pointed out by Nerhagen et al. (2005), the most recent costs agreed upon to achieve implementation of the Gothenburg Protocol are not included in the estimates by Vermoote and De Nocker (2003), making them underestimates in that sense. Vermoote and De Nocker give 350 € ha^{-1} year^{-1} as an upper margin for marginal WTP, including all proposed emission reduction programs in Europe. However, measures intended to reduce acidification are a subset of these programs since they also include measures to protect ecosystems from both eutrophication and the impacts of ozone, as well as protect human health.

Table 2. *Studies estimating benefits of recovery of freshwater ecosystem services affected by air pollution.*

Ecosystem services	Context	Valuation method	Country	Study
Provisioning services	Commercial fisheries	Production function approach	Canada	Hough and J.E. Hanna Associates (1982)
	Commercial fisheries	Contingent valuation (relationship between acid deposition levels and the willingness of fishermen to pay to avoid its impacts using the Air Quality Valuation Model)	Canada	Chestnut et al. (1999)
	Fish stocks	Contingent valuation/Travel cost method /Cost based method	Norway	Navrud (1989), Navrud (1993a), Navrud (1993b), Navrud (1993c), Navrud (2001a), Strand (1981)
	Damage caused to fish stocks	Cost based method	US	Menz and Driscoll (1983)
Regulating services				
Cultural services	Stream pH affecting property values	Hedonic Price method	Canada	Epp and Alani (1979)
	Recreational fishing	Cost based method	Canada	Bourassa et al. (2004)
	Recreation and swimming in the lake with only minor usage for fishing	Contingent valuation	Finland	Iivonen et al. (1995)
	Recreational fishing	Contingent valuation	Norway	Navrud (2001b), Navrud (1989)

	Recreational fishing, existence values	Benefit transfer	Sweden	Bostedt et al. (2010)
	Recreational fishing, salmon	Cost based method	UK	Milner and Varallo (1990)
	Recreational fishing, salmon	Hedonic price method	UK	MacMillan and Ferrier (1994)
	Recreational fishing	Travel Cost method	US	Violette (1985), Mullen and Menz (1985), Morey and Shaw (1992), US-EPA (1999)
	Recreational fishing	Production function approach	US	US-EPA (1999)
Supporting services	Biodiversity recovery	Contingent valuation	UK	Macmillan et al. (1996), ECOTEC (1993)
Other	Benefits of liming	Contingent valuation	UK	Bateman et al. (2005)

5. Case study Tovdal River, Norway – valuation of freshwater ecosystem services affected by acidification

5.1 Acid deposition causes loss of ecosystem services

The Tovdal River is a major river in southernmost Norway and runs north-to-south from its headwaters in the uplands above 1000 m above sea level to the Topdalsfjord at the coast at Kristiansand (***Figure** 7*). The catchment area is 1863 km^2 and underlain predominantly by Precambrian granitic and gneissic bedrock, with thin and patchy moraine of the same lithology. The higher lying areas of the catchment are characterised by alpine, heathland and peaty soils. Much of the lower parts are forested with pine, spruce and birch. There is very little farming, industry or habitation in the catchment and it has not been developed for hydropower significantly. The river basin is highly sensitive to acid deposition.

Figure 7. *The Tovdal River at Boen waterfall. Photo: Frode Kroglund*

Decades of chronic acid deposition starting as early as the late 1800's have caused extensive and severe acidification of freshwaters in the Tovdal River basin. The Tovdal catchment contains nearly 300 lakes, the majority of which were highly acidified by the 1970s and had lost their native fish populations (mainly brown trout) in the period 1940-1980.

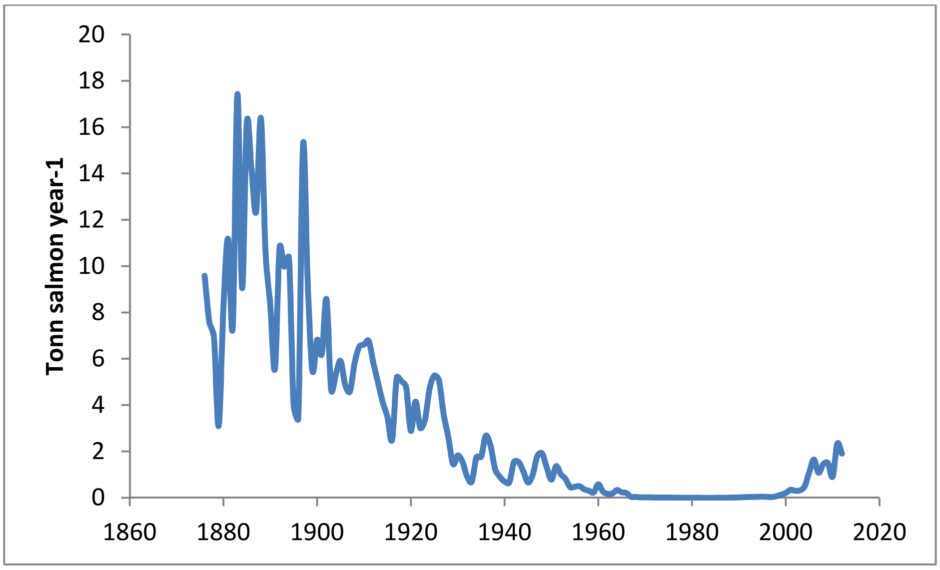


Figure 8. *Salmon catch statistics for the Tovdal River 1876-2012. Liming started in late 1995. Source: Statistics Norway*

In the 1800's and up to about 1950 the Tovdal River was one of the best salmon rivers in southern Norway. In the period 1876-1885 the annual catch was 9.5 tonnes (***Figure 8***). But just like several other major salmon rivers in southernmost Norway, the catch began to decline in the early 1900's and by the 1950's only a few hundred kg were caught. By 1970 the entire population was wiped out.

Figure 9. *The Tovdal River at Boen waterfall. Photo: Frode Kroglund*

Figure 10. *The salmon fishery in the Tovdal River has a long history. The visit by King Christian IV in 1631 is commemorated by the inscription in a rock near the famous Boen waterfall. Photo: Dag Matzow*

Figure 11. *The «good old days» at the Tovdal River. Mr William Radcliffe at the Boen waterfall in the 1920's. In 1924 he and this companion Mr Harold Wilson caught 1352 salmon with two rods during 51 days of fishing. Source: Haraldstad and Hesthagen (2003).*

The loss of the salmon meant the loss of all the activities dealing with salmon fisheries. The English lords no longer came to fish, the local landowners and fishermen lost a valuable source of income, and the local community was affected.

Liming as a mitigative measure
Beginning in late 1995 the river has been limed to raise the pH such that water quality is adequate for salmon. Salmon have begun to return to the river. Liming has also benefited the brown trout population in the river, as well as macroinvertebrates, macrophytes and other organisms (DN, 2010).

Liming costs 1996-2000 were about NOK 3 million/yr (420 k€/yr).

Figure 12*. English sport fishermen together with locals about year 1900. Note the salmon hanging on the wall in the background. Source: Haraldstad and Hesthagen (2003)*

5.2 Possible methods for valuation of freshwater ecosystem services affected by acidification in the Tovdal River

The freshwater ecosystem services affected by acidification in the Tovdal River, could be valued using more or less comprehensive methodologies as presented in chapter 3. In a river like Tovdalselva, where there are a lot of historical data on the environmental status, mitigation measures and their costs, cost-based methods could be a good alternative for valuation. Cost-based methods involve observing the real costs (including opportunity, prevention, avoidance, replacement, mitigation costs and damage costs) incurred by stakeholders in adapting to changes in ecosystems. These methods assume that the expenditures involved in preventing/mitigating or replacing losses of ecosystem services represents what people are willing to pay for the ecosystem services. However, WTP could be both lower and higher than these costs.

One approach to quantify the freshwater ecosystem services affected by acidification in the river Tovdalselva, is to base this on the total costs of measures/mitigation and assess whether the benefits are proportional to the costs. Using the present value method, one can calculate what the benefits or willingness-to-pay has to be to equal or exceed the cost of mitigation. These numbers can be compared with willingness-to-pay studies carried out in nearby or relevant other water systems to see if the results are similar. When cost data are available and primary valuation studies of benefits do not exist, this approach is useful to evaluate the profitability in terms of benefits and costs. However, this requires that the willingness-to-pay is about the same in the source site as in the target site.

6. Knowledge gaps

Although environmental economic research has come a long way, there are still large gaps in our understanding of the nature and value of acid deposition impacts. Despite the fact that some economic-evaluation modelling capacity currently exists (e.g. the Environment Canada's Air Quality Valuation Model) (Bourassa et al., 2004), economic-evaluation models for acid deposition do not adequately account for environmental benefits resulting from abatement. Hence, future research efforts should focus on quantifying the benefits and costs associated with acid deposition effects on forest growth and productivity, recreational fishing, wildlife consumption and biodiversity.

7. Some concluding remarks

The implementation of measures in Europe and North America to reduce the emissions of air pollutants and thereby reduce acid deposition shows that society as a whole places a higher value on the terrestrial and aquatic ecosystems than the cost of implementing these measures. It is the total value of the ecosystem services provided, including aesthetic and ethical aspects that are not readily quantified in terms of economic value as well as the more-easily quantifiable services such as loss of commercial fisheries, that have been judged to be worth the costs of emission reductions. This has not always been the case, however. An editorial in 1977 in the well-respected scientific journal Nature called acid rain "a million dollar problem with a billion dollar solution" (Anon., 1977), referring to the loss of salmon on the one hand and the cost of installing scrubbers for sulphur removal at power plants on the other hand. In Europe the signing of the First Sulphur Protocol of the LRTAP Convention in 1986 marked a turning point –countries now agreed that reducing emission was worth the cost in terms of protection of the environment (i.e. protecting ecosystem services). Similarly in the United States the signing of the Clean Air Act Amendments in 1990 entailed a commitment to reducing acid deposition, and thus the first step in protecting ecosystem services. Also in Canada similar legislation was instigated to reduce acid deposition and impacts on ecosystems.

Nevertheless, it appears that there has never been a thorough analysis of the total costs and benefits of reducing acid deposition in terms of ecosystem services.

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10. Reports and publications from the ICP Waters programme

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Reports before year 2000 can be listed on request.

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