

European Nitrogen Assessment

Chapter 22: Costs and benefits of dealing with nitrogen in the environment

Supplementary Material: Background information on costing nitrogen

Authors: van Grinsven, H., Ing-Marie Gren, Kevin Hicks, Roald Dickens, Jaap Willems, Mette Termansen, Gerard Velthof, Rob Alkemade, Mark van Oorschot

22.S1. Dealing with uncertainties

Cost Benefit Analysis (CBA) should have a central role in the development, design and evaluation of policies that influence the nitrogen cycle. In an idealised system it would be possible to estimate all the impacts of a given action and then value all these outcomes. The policy decision would then be relatively simple matter of optimising net social welfare against any given constraints, such as tax revenues. These decisions should then be evaluated after implementation to ensure that they are delivering the expected benefits and that there are no unintended additional costs.

In practice, however, such a comprehensive approach is unlikely to ever be fully operationalised. Therefore it is necessary to consider how best to develop policy in such a world. This section therefore considers one of the obstacles to a comprehensive evaluation approach, i.e. the vast range of potential informational issues. These include both information gaps and uncertainty around the available information. This section will explain the challenge and illustrate some of the more prevalent approaches to this issue.

Considering information gaps first, in a simple example of a policy to avoid climate change this policy could only be assessed if two key pieces of information are available: the link between the action policy and carbon emissions; and the value of prevented climate change. Unless reasonable estimates of both these factors are available, it is not possible to evaluate if the benefits of the policy would outweigh its costs. Fortunately, a great deal of the necessary data on both sides of the cost-benefit equation has already been collated for many air and water pollution issues, though problems remain. For example, in the field of air pollution assessment, little progress has been made with respect to the characterisation and quantification of impacts of nitrogen on ecosystems beyond identification of areas that are at some risk through exceedance of the critical load. In cases where quantified benefits do not exceed costs, or the benefit:cost ratio does not pass some threshold, 'switching point' analysis may be used to see how much larger benefits would need to be to justify the policy under investigation. Once this is known, it is possible to consider qualitatively whether the effects that remain unquantified are perceived to be sufficiently large that the switching point would be met. This process is clearly made easier the more information is available on the unquantified effects.

Uncertainty can become a major factor if the perceived range covers the switching level between a policy being cost beneficial or not. Uncertainties are of several types: (i) data uncertainties which can often be described using statistical distributions and dealt with accordingly, (ii) modelling assumptions that may be approached using sensitivity analysis, and (iii) biases resulting from (e.g.) a lack of data. There is a

tendency to regard uncertainties as affecting estimates of benefit more than estimates of cost. This view arises from the perspective that the costs of individual and well defined abatement measures can be described with a reasonable level of accuracy, though there is a tendency for actual costs of specific measures to be lower through improvements in efficiency and identification of other cost savings once measures start to be widely deployed. When a policy permits flexibility (e.g. through setting national emission ceilings) uncertainties in abatement costs become much more significant as experience shows that the response of industry (etc.) can be different to that originally envisaged. As it is clearly in the interests of affected parties to minimise costs once a policy is in force, the tendency is for this source of bias to exaggerate costs.

For CBA in relation to air quality policy, methods for addressing uncertainty have been refined in recent years through work by Rabl *et al.* (2005) and Holland *et al.* (2005) under the EC's CAFE programme. The latter treats statistical uncertainties in benefit estimates using Monte Carlo techniques, combined with sensitivity analysis on key parameters (estimated costs of specific scenarios and valuation of mortality). A framework has also been proposed for dealing with the unquantified elements in the analysis, considering which are likely to be of most significance and in what direction their omission is likely to bias results.

One particular example would be the impact pathway approach to evaluating air quality. This approach follows the logical progression from pollutant emission through atmospheric dispersion to exposure and on to a range of end-points that are valued (such as health outcomes). While this is a robust approach to evaluate these impacts, there are uncertainties at each stage leading to potentially a vast range of estimates. If, for example, we assume a small margin of tolerance of $\pm 15\%$ at each of the four stages (emission, dispersion, dose response and valuation), the high estimate would be almost three and a half times the low estimate. This substantial gap means that there is a strong possibility that the evidence would reach an indeterminate result. There are two problems with such an approach. First the failure to distinguish which parts of the analysis are most robust and which are most uncertain and second, the calculation of extreme positions that fail to recognise the tendency for uncertainties to cancel one another out to some extent. Whilst the extremes so calculated may be theoretically plausible they also have a low probability of occurrence compared to more central values.

Given the informational issues, the subsequent challenge is to try and balance the objective to support decision making and need to reflect the rich, complex, and sometimes contradictory, evidence. Given such a broad range of potential outcomes it may be very challenging to balance these two competing goals.

A wide range of techniques are commonly used to try and allow the communication of a clear message while also providing the necessary context. The most common historic approach has been to develop a central point or range for the proposal and to provide additional supporting detail on the possible and likely uncertainties and the importance of them through sensitivity analysis. Although this is an extremely useful approach, there may be concerns that the uncertainty analysis may not have the attention it deserves. Other approaches look to bring in the key sensitivities within a central range. Two classic approaches include the use of fan diagrams and Monte Carlo analysis. Fan charts are particularly strong to reflect uncertainties over time. Fig. 22.S1.1 shows an example of a fan diagram, showing an increasing range for the estimate price increase until 2011. An alternative approach has been through the use of Monte Carlo analysis that incorporates the uncertainties surrounding the

information to provide estimates of the likelihood that the benefits exceed the costs. Fig. 22.S1.2 provides an example of a Monte Carlo analysis of a particular measure in the UK Air Quality Strategy. The figure clearly shows that there is only a tiny probability that the overall cost (once benefits are factored in) exceeds zero.

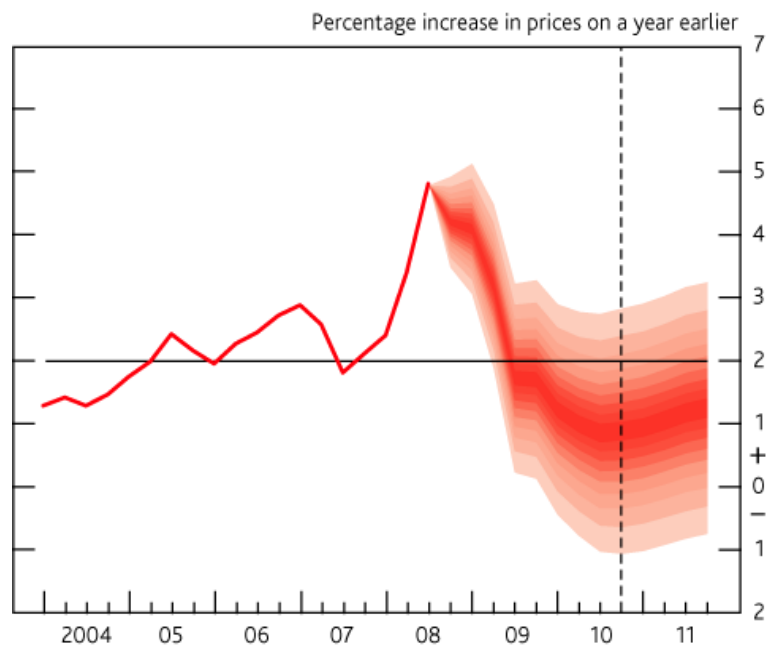


Figure 22.S1.1 Fan diagram for prices increase in the UK until 2011. Source: Bank of England Inflation Report November 2008, <http://www.bankofengland.co.uk/publications/inflationreport/index.htm>.

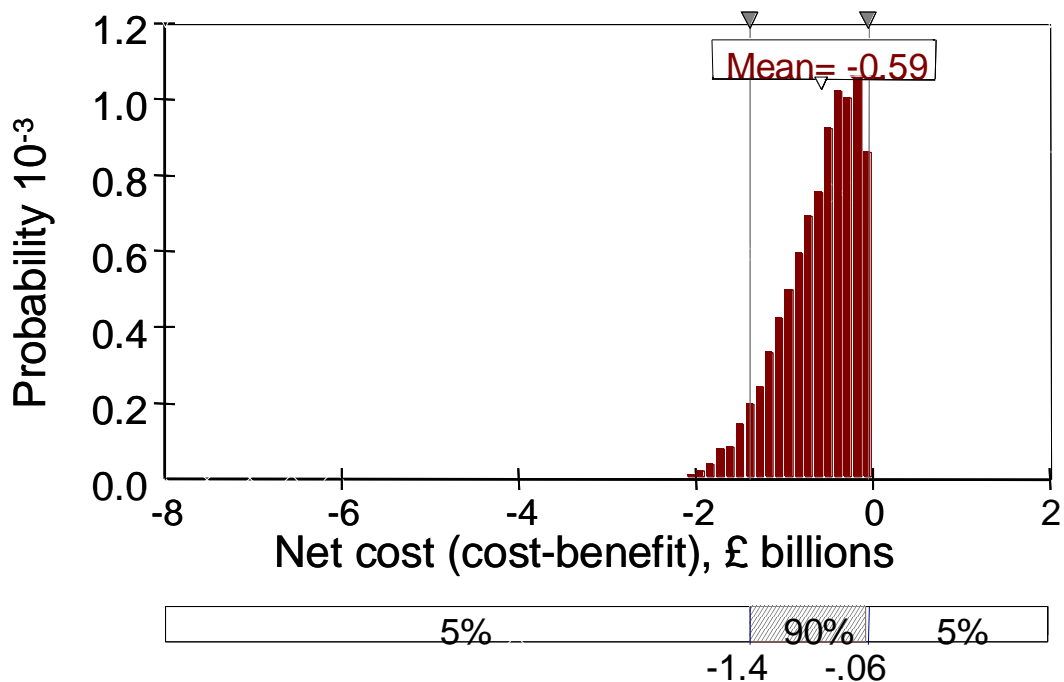


Figure 22.S1.2 Monte Carlo results of a particular measure in the UK Air Quality Strategy (2007). Source: <http://www.defra.gov.uk/environment/airquality/panels/igcb/index.htm>.

22.S2. Agricultural benefits

Winter wheat

Winter wheat is the major arable crop in Europe using about 25% of agricultural area and of total N-fertilizer use in Europe (EU27), and was therefore selected to provide some more information on variation and uncertainty of nitrogen benefits. The used data represent a wide range of conditions, rotations and setups of field trials. The net unit benefit of nitrogen (UBoN: $(Y_{opt} - Y_o) \cdot P_C / N_{opt} - P_N$) ranges from 0,4 euro/kg_N in SE Europe to 2,7 euro/ kg_N in NW Europe (Table 22.S2.1).

Table 22.S2.1 Characteristics of observed response of grain yield of wheat to nitrogen fertilizer using quadratic response models. Y_{max} is maximum yield at N-rate of N_{max} . Y_o is the grain yield at zero N-rate. Y_{opt} is economically optimal yield at N-rate = N_{opt} . N_{EFMA} and Y_{EFMA} are country average actual yields and N-rates (EFMA, 2008). Unit benefit of N (UBoN) is the slope of the chord at N_{opt} and based on a wheat price of 125 euro/ton and a CAN-fertilizer price of 0.8 euro/kg_N.

	Country	Soil	Y_{max}	N_{max}	Y_o	N_{opt}	Y_{opt}	N- EFMA	Y- EFMA	UBoN at N_{opt}
			ton/ha	kg/ha	ton/ha	kg/ha	ton/ha	kg/ha	ton/ha	euro/kg
1	Germany	Sandy loam	9,3	201	4,6	173	9,2	125	7,2	2,5
2	Germany	n.r.	9,7	242	5,5	197	9,5	125	7,2	1,8
3	Spain	clay	2,6	128	1,8	61	2,3	86	3,1	0,4
4	UK	sandy loam	8,6	204	5,2	165	8,5	192	7,8	1,7

5	Portugal	clay	3,5	223	1,6	137	3,2	90	2,4	0,6
6	Switzerland	n.r.	7,1	258	3,1	205	6,9	150	5,9	1,5
7	Denmark	sandy loam	8,0	167	4,3	142	7,9	150	7,2	2,3
8	England	loamy sand	8,4	235	4,2	193	8,3	192	7,8	1,9
9	England	clay-loam	10,7	273	3,8	238	10,6	192	7,8	2,7
10	Germany	n.r.	9,8	241	4,5	206	9,7	125	7,2	2,3

References

1. Henke J, G Breusted, K Sieling and H Kage (2007). Impact of uncertainty on the optimum nitrogen fertilization rate and agronomic, ecological and economic factors in an oilseed rape based crop rotation. *Journal of Agricultural Science*, 145: 455-468.
2. Lammel J, F Brentrup & J Küsters (personal communication)
3. Lopez-Bellido, RJ, Lopez-Bellido L, Benitez-Vega J & FJ Lopez-Bellido (2007) Tillage System, Preceding Crop, and Nitrogen Fertilizer in Wheat Crop: II. Water Utilization. *Agron. J.* 99: 66-72.
4. Lord, EJ & RDJ Mitchell (1998) Effect of nitrogen inputs to cereals on nitrate leaching from sandy soils. *Soil Use and Management*, 14:78-83.
5. Carvalho M & G Basch (1996) Optimisation of nitrogen fertilisation, Relationship between precipitation and expected yields. *Fertilizer Research* 43: 127-130
6. Charles, R , O Jolliet, G Gaillard & D Pellet (2006). Environmental analysis of intensity level in wheat crop production using life cycle assessment. *Agriculture, Ecosystems and Environment* 113 (2006) 216–225.
7. Nyord T & T Birkmose (2006) Precise Injection of Slurry between Rows of Winter Wheat. Paper Symp. Precision Livestock Farming 2007, Skiathos, Greece. 3. - 06. june 2007.
- 8+9. Dampney P, C Dyer, G Goodlass & B Chambers (2006). Crop responses to different N fertiliser materials. Component report for Defra report NT 2605. WP1a Crop responses.
10. Busenkell, J (2004) Beurteilung von Agrarumweltmaßnahmen in Nordrhein-Westfalen und Rheinland-Pfalz. Diss Rheinischen Friedrich-Wilhelms-Universität zu Bonn. Übersicht 18 (Beispiel Winterweizen).

Oilseed Rape

Oilseed is an emerging crop but the market and price for oilseed is uncertain because of current developments in EU policies for climate and biofuels. Values of UBoN (Table 22.S2.2) range between 0.3 and 2.3 euro/kg and therefore are somewhat lower than for winter wheat in view of lower yield and weaker response to N.

Table 22.S2.2 Characteristics of observed response of grain yield of winter oilseed rape to nitrogen fertilizer using quadratic response models. Y_{max} is maximum yield at N-rate of N_{max} . Y_o is the grain yield at zero N-rate. Y_{opt} is economically optimal at N-rate = N_{opt} . N_{EFMA} and Y_{EFMA} are country average actual yields and N-rates. Unit

benefit of N (UBoN) is the slope of the chord at N_{opt} and based on an oilseed price of 220 euro/ton and a CAN-fertilizer price of 0.8 euro/kg_N.

	Country	Soil	Ymax	Nmax	Yo	Nopt	Yopt	N- EFMA	Y- EFMA	UBoN- Yopt
			ton/ha	kg/ha	ton/ha	kg/ha	ton/ha	kg/ha	ton/ha	euro/kg
1	England	n.r.	4,2	223	2,3	174	4,1	195	3,2	1,4
2	England	clay/s loam	4,1	308	1,9	229	4,0	195	3,2	1,2
3	England	n.r.	3,7	282	1,8	207	3,6	195	3,2	1,1
4	Scotland	n.r.	5,5	403	4,4	153	5,1	195	3,2	0,2
5	Germany	sandy loam	4,6	246	1,8	207	4,6	165	3,6	2,2
6	Germany	sandy loam	5,4	220	2,7	187	5,3	165	3,6	2,3
7	Germany	sandy loam	4,5	221	2,7	172	4,4	165	3,6	1,4
8	Poland	sand	3,7	129	2,2	108	3,7	133	2,7	2,1
9	Lithuania	sandy loam	2,5	179	1,9	81	2,3	107	1,6	0,3

References

- Richards, IR (2000) Energy balances in the growth of oilseed rape for biodiesel and of wheat for bioethanol. Levington Agriculture Report for the British Association for Bio Fuels and Oils (BABFO).
- Berry PM & JH Spink (2009) 'Canopy management' and late nitrogen applications to improve yield of oilseed rape HGCA Project Report No. 447, March 2009.
- Defra (2009) LINK project – N efficiency of OSR – trial results.
http://www.seedquest.com/news.php?type=news&id_article=2592&id_region=10&id_category=331&id_crop=
- Booth , E (1998) Oilseed rape varieties for the millennium. HGCA Agronomy road show 1998.
http://www.hgca.com/document.aspx?fn=load&media_id=1055&publicationId=681
- Henke J, G Breusted, K Sieling and H Kage (2007). Impact of uncertainty on the optimum nitrogen fertilization rate and agronomic, ecological and economic factors in an oilseed rape based crop rotation. *Journal of Agricultural Science*, 145: 455-468.
- Sieling K & H Kage (2008) The potential of semi-dwarf oilseed rape genotypes to reduce the risk of N leaching. *Journal of Agricultural Science* 146: 77–84
- Rathke GW, O Christen & W Diepenbrock (2005) Effects of nitrogen source and rate on productivity and quality of winter oilseed rape (*Brassica napus* L.) grown in different crop rotations. *Field Crops Research* 94:103–113
- Barlóg, P & W Grzebisz (2004). Effect of timing and nitrogen fertilizer application on winter oilseed rape (*Brassica Napus* L.) I. Growth dynamics and seed yield. *J. Agronomy and Crop Science* 190: 305-313.
- Sidlauskas, G & S Bernotas (2003) Some factors affecting seed yield of spring oilseed rape (*Brassica Napus* L.) *Agronomy Res.* 1(2): 229-243

Dairy - grass and milk

The share of use of roughage and feed concentrates for the dairy sector in Europe varies strongly; with high shares of roughage in regions with large areas of productive grassland and low land prices (N- Europe, Ireland) and lower shares in regions with intensive dairy production but high land prices (Netherlands, Flanders) or low productivity (semi-arid).

Using yield response data for grassland in the UK, the Netherlands and Flanders UBoN was found to range between 1,2 and 3,3 euro/kg_N (Table 22.S2.3) and therefore similar to values for wheat in NW European countries. UBoN values for production of roughage for silage for these four cases range between 0,6 and 2 euro/ kg_N using a herbage price of 100 euro/ton.

Table 22.S2.3 Characteristics of observed response of milk yield to nitrogen fertilizer application to grassland for cutting only, and grazing (*) using quadratic response models for herbage production and linear response of milk yield to effective N in total feed (efficiency of N-fertilizer for milk production is 40% as compared to N in feed concentrate; see also paragraph 3.6). Unit benefit of N (UBoN) is the slope of the chord at N_{opt} and using a CAN price of 0,8 euro/kg_N and a milk price of 290 euro/ton.

	Country	Soil	Ymax ton/ha	Nmax kg _N /ha	Y ₀	N _{opt}	Y _{opt}	N- EFMA	UBoN- Y _{opt}
1	UK		6,5	509	2,6	418	6,4	68	1,8
2	Netherlands		10,3	508	3,8	453	10,2	171	3,3
3	Flanders		10,1	493	7,4	368	10,0	132	1,2
4	Flanders*		8,2	317	6,0	253	8,1	132	1,6

References

1. Shiel, RS, AMA El Tilib & A Younger (1999) The influence of fertilizer nitrogen, white clover content and environmental factors on the nitrate content of perennial ryegrass and ryegrass/white clover swards *Grass and Forage Science*, 54: 275-285.
2. Schils RLM, HG van der Meer, AP Wouters, JH Geurink & K. Sikkema (1999) Slurry following surface application to grassland. *Nutrient Cycling in Agroecosystems* 53: 269–280.
- 3+4. Nevens F & Reheul D (2003) Effect of cutting or grazing grass swards on herbage yield, nitrogen uptake and residual soil nitrate at different levels of N fertilization. *Grass & Forage Science*, 58: 431-449.

22.S3. Valuation of the impacts of biodiversity

The monetization procedure in the study Cost of Policy Inaction (COPI; Braat and ten Brink (eds), 2008) uses the conceptual link between biodiversity (hereafter addressed as Mean Species Abundance; MSA) and ESS; where changes in the economic value of ESS are the result of a change in MSA (e.g. related to N) and a change of the area of ecosystems due to land use change (Figure S22.3.1).

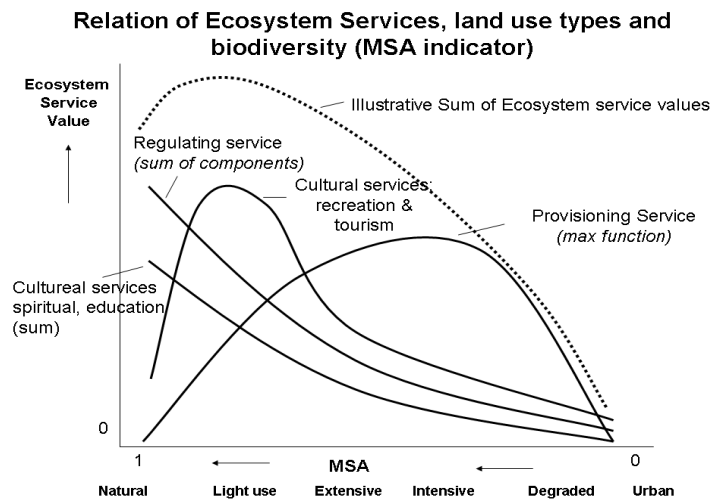


Figure 22.S3.1. Conceptual relationship between biodiversity and ecosystem services (Taken from Braat and ten Brink (2008)).

It was estimated in COPI that the current global loss of welfare due to loss of land-based ESS is 50 billion euro per year, and that this loss would amount to 14 trillion euro per year (7% GDP loss) in 2050 if no policy action would be taken. The calculated increase of economic loss by a factor of nearly 300 between 2000 and 2050 is partly due to loss of ESS and partly to “multipliers” related to increase of income and population, presumably increasing demand and “Willingness To Pay” for ESS. COPI uses a variety of cost data, but in fact only 56 ESS values were used from 5 references. The economic cost due to loss of ESS for Europe in 2050 was estimated at 1,1 trillion euro/year.

Share of nitrogen in loss of biodiversity and ecosystem functions

The contribution of nitrogen deposition to the overall loss of abundance of species (MSA) in terrestrial Europe in 2000 was estimated to be 2% (MNP, 2008), and is expected to decrease in the future due to success of nitrogen policies. The major factors causing MSA-loss are land use by agriculture, infrastructure and settlement and fragmentation (MSA loss is essentially driven by land use change). However, within the remaining natural areas the relative contribution of nitrogen to loss of MSA in Europe was estimated at about 30% and much larger than the N-share to the total loss of biodiversity (personal communication Rob Alkemade). The greater importance of nitrogen for MSA in natural systems is also illustrated by empirical data from 22 papers on the experimental addition of nitrogen (Alkemade et al. 2009).

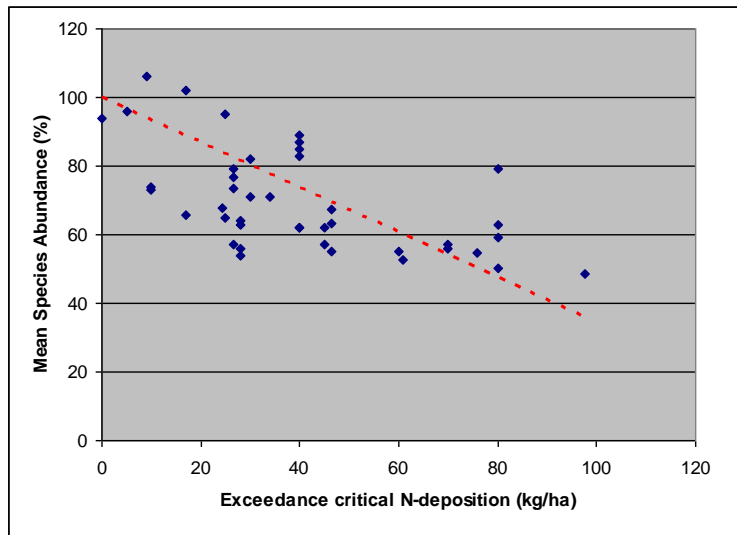


Figure 22.S3.2 MSA and regression lines for exceedance of critical nitrogen loads. Each data point represents a single study for grassland, forest or arctic systems

Figure S22.3.2 shows that 100 kg of exceedance will cause an MSA loss of about 50%, equivalent to 0,5% MSA loss per kg of N, but Alkemade et al. (2009) suggest a logarithmic decrease, where MSA loss decreases with increasing exceedance.

Although progress has been made to quantify the economic value of ecosystems using ESS, there are hardly any studies providing quantitative information about the role of N in ESS loss, both in terms of dose response and economic value, and that also have some representative value for ecosystems in Europe. Table 22.S3.1 shows examples of how nitrogen in the environment can influence ESS and also serves to illustrate how many of these services are intimately linked to biodiversity. Functional changes (e.g. increased soil N cycling and nitrate leaching and increased decomposition rates and loss of stored carbon) may be caused directly by changes in nitrogen addition and/or indirectly through structural changes, making assessment difficult and complex.

Table 22.S3.1 Examples of the Effect of Nitrogen on Ecosystem Services (after Hicks *et al.*, 2008)

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

22.S4. Decreasing the release of N₂O

Basically, there are two strategies in farming systems to decrease N₂O emissions from agriculture (Oenema *et al.*, 2001):

- Balanced N fertilization, i.e. increasing the N use efficiency concomitant with a lowering of the total N input with reduction in yield, and
- Decreasing the release of N₂O per unit N from the processes nitrification and denitrification.

Increase of N-use efficiencies

Improving the N use efficiency reduces both the direct N₂O emission from soils and the indirect N₂O emissions associated with ammonia emission and nitrate leaching. Measures to increase the N use efficiency in crop production systems, include adjustment of N application rate, method, and timing to the crop demand, use of soil and plant testing as basis for N fertilization, proper manure management (including grazing systems), and accounting for mineralization of organic N. Adjustment of crops in rotation and growth of winter crops are also options to increase N use efficiency. More general measures that optimize the growing conditions may increase N use efficiency, including application of other nutrients, proper soil cultivation, liming, proper water management (drainage and irrigation) and good crop protection against diseases and weeds. In general, there are no net costs or costs are low of these options to increase N use efficiency, because they result in a higher yield and/or less use of mineral N fertilizer. In an assessment of the global potential to mitigate greenhouse gas emissions, Smith *et al.* (2008) estimated the costs of improved nutrient management (i.e. options to improve the use of applied N) at 5 US\$ per ha cropland and of agronomy (i.e. agronomic practices to increase yields, such as changes in crop rotations) at 20 US\$ per ha cropland.

Decreasing the release of N₂O per unit N

Nitrous oxide is produced during the microbial processes denitrification and nitrification, but highest emissions are generally caused by denitrification. There are several factors affecting the rate of denitrification and nitrification and the production of N₂O per unit N transformed (i.e. the N₂O/N₂ ratio of the end products of denitrification). Measures to reduce N₂O emission have to focus on avoiding application of N during these conditions or to change the conditions to conditions less favourable for N₂O production (Oenema *et al.*, 2001). Various measures are available to decrease the release of N₂O per unit N from the processes nitrification and denitrification:

- Using ammonium based fertilizer (including urea) instead of nitrate fertilizer during wet conditions may significantly reduce N₂O emission (Clayton *et al.*, 1997; Dobbie & Smith, 2003; Jones *et al.*, 2005; 2007, and Velthof *et al.*, 1997). There are no significant costs of using of ammonium based fertilizer instead of nitrate based fertilizers, but emissions of ammonia will increase.
- Available carbon is an important energy source for denitrifying bacteria (Tiedje, 1988). Avoiding conditions with high contents of available carbon and nitrate in the soil is an option to decrease N₂O emission. This includes measures as no manure application to a soil with high nitrate concentration, no application of

nitrate containing fertilizer to a manured soil or to a soil with crop residues, and the careful use of nitrate containing fertilizers in soils with a high organic carbon contents. The costs of this type of measures are low, because they are based on correct timing of N application and choice of fertilizer type.

- Nitrification inhibitors delay the transformation of ammonium to nitrate (and possible denitrification of the produced nitrate). The use of nitrification inhibitors in agricultural systems in which ammonium based fertilizers or manures are used may reduce N₂O emission (Dobbie & Smith, 2003; Mejjide *et al.*, 2007). The costs of a fertilizer containing nitrification inhibitor is about a factor 1.5 to 2 higher than a common ammonium based fertilizer.
- Denitrification is hampered by the presence of oxygen (it is an anaerobic process). Nitrification is an aerobic process (needs oxygen), but N₂O production during nitrification occurs at low oxygen concentrations in the soil. Enhancing aeration of soils by proper drainage, irrigation and soil tillage and avoiding application of N during wet conditions are strategies to reduce N₂O emission from soils. The cost of this type of measures is generally low. Avoiding application of N during wet conditions does not have costs and costs related to tillage amount to 5 US\$ per ha cropland Smith *et al.* (2008). However, a construction a system for drainage of agricultural field will be costly for the farmers.
- Crop residues incorporated in soil are a potentially important source of N₂O (Velthof *et al.*, 2002; Vinther *et al.*, 2004). The N₂O emission is much higher for green residues (e.g. residues from vegetables and sugar beet) than from straw. Removal of crop residues from the field (e.g. for animal feed, bedding materials, composting, digestion, fuel etc.) is an option to decrease N₂O emission. The costs are relatively high compared to other options to mitigate N₂O emission, because it requires equipment to collect the residues, and additional costs for handling and storage of the residues.
- Winter crops or catch crops reduce the nitrate content of the soil in the winter, by which N₂O emission may be reduced. In spring, winter crops are ploughed and the N is then released during the growing season. Thus reduces the amount of required fertilizer and the related N₂O emission. The costs are related to soil tillage and seed, and are higher than costs related to improved N management.
- The anaerobic conditions in storage of liquid manure and slurry limit nitrification, by which little N₂O is lost (Sommer & Petersen, 1997). However, in storage of solid manures, mixing may cause aeration of manure, which favours production of N₂O (Groenestein & van Faassen, 1996; Sommer *et al.*, 2004; Chadwick, 2005). Also composting of manures may result in N₂O emission. Proper mixing of the manure may decrease N₂O emissions from solid manure systems (Sommer, 2001). The costs of these measures are relatively low. The largest effect is obtained when solid manure systems are changed into anaerobic storage systems. This has large costs, because it demands for changes in manure collection, handling, and storage systems. The liquid based systems may have high higher NH₃ and CH₄ emissions than solid manure based system, which partly counterbalances the decrease in N₂O emission. A high share of cattle farming based on solid manure systems (i.e.

more than 50% of the cattle manure is collected as solid manure) are found in central European countries and South European countries (Source: GAINS model). The share of solid manure systems is generally (much) lower in pig farming systems than in cattle farming systems). In Italy, United Kingdom, Ireland, Poland, Latvia and Bulgaria the share of solid manure systems in total pig farming is relatively high.

22.S5. Integrated nutrient management of the Baltic Sea: description of modelling and data retrieval

A mathematical programming model is used for calculating costs of nutrient – nitrogen and phosphorus - reductions to the Baltic Sea. Due to lack of data, in particular on the impact of nutrient reductions on biological parameters such as water transparency and cod reproduction it is not possible to relate benefits to nutrient reduction. Instead, costs are compared with expected benefits obtained under the Helcom Baltic Sea Action Plan, which allocates nutrient reductions among the nine riparian countries (Helcom, 2007). Data on associated benefits are obtained from contingent valuation studies on willingness to pay for a ‘healthy’ Baltic Sea carried out in mid 1990s in Sweden and Poland (Markowska & Zylicz, 1999). The results from these studies were transferred to the remaining riparian countries. Updated results are used in this study (Söderqvist and Hasselström, 2008).

When calculating cost effective nutrient reductions, the entire drainage basins of the Baltic Sea is divided into 24 drainage basins with nutrient loads into one of the marine basins. Nutrient transports from sources and costs of abatement measures are calculated for each of these drainage basins, which are briefly presented in this chapter. Unless otherwise stated, all data and calculations are found in Gren *et al.* (2008). Nutrient loads to the Baltic Sea are, for all emission sources, calculated by means of data on emissions, which is sufficient for sources with direct discharges into the Baltic Sea, such as industry and sewage treatment plants located by the coast and air deposition. For all other sources further information is needed on the transformation of nutrients from the emission source to the coastal waters. This requires data on transports of airborne emissions among drainage basins, leaching and retention for all sources with deposition on land within the drainage basins, and on nutrient retention for upstream sources with discharges into water streams. Nitrogen loads are therefore divided into three main classes; airborne emissions, agricultural loads, and discharges of sewage from households and industry. Phosphorus loads are classified into the same categories with the exclusion of airborne emissions.

Airborne emissions include nitrogen oxides and ammonia which are deposited in the drainage basins and directly on the Baltic Sea. This study includes all air deposition on land within the drainage basin, which originates from countries within and outside the drainage basin, which can be affected by abatement measures in the drainage basin or coastal waters. This is not the case for direct air deposition on the open sea originating from non-riparian countries, which accounts for approximately 5 per cent of total load to the Baltic Sea (Gren *et al.*, 2008). The airborne emission gives rise to deposition directly on the Sea and also indirectly through deposition on land, which is transported by soil and water into the Baltic Sea. Calculation of indirect air deposition and loads from agriculture is made by data on airborne transports of nitrogen and ammonia among countries, deposition on land and on leaching from soil and retention in water transports to the Baltic Sea.

The contribution of nutrient loads to the Baltic Sea from arable land is calculated in the same way as for indirect air deposition. Deposition of nutrients on arable land then includes manure and fertilisers. Estimation of discharges of nutrients from households

are based on annual emission per capita in different regions, and on connections of populations to sewage treatment plants with different cleaning capacities. It is assumed that remaining nutrients from households and industry in the drainage basins are discharged into water streams, and the final deposition into the Baltic Sea then depends on nutrient retention. Given all assumptions, the calculated total nutrient loads of approximately 830 kton of nitrogen and 40 kton of phosphorus, which come relatively close to the estimates obtained in Helcom (2004).

The cost minimization model includes a number of different abatement measures which either reduce nutrient loads from sources or act as sinks for nutrients. Examples of the former are selective catalytic reductions at combustion sources, livestock reductions, and decreases in use of nitrogen fertilisers. Land use changes such as construction of wetlands and grass land provide examples of measures reducing downstream nutrient transports. For each abatement measure, costs are calculated which do not include any side benefits, such as provision of biodiversity by wetlands. Furthermore, abatement measures located in the drainage basins may have a positive impact on water quality, not only in the Baltic Sea, but also in ground and surface waters. However, such data on side benefits is not available for the included abatement measures. This implies an overestimation of abatement costs of measures implemented in the drainage basins. On the other hand, the cost estimates do not account for eventual dispersion of impacts on the rest of the economy from implementation of the measure in a sector, such as eventual increase in prices of inputs of a simultaneous implementation of improved cleaning at sewage treatment plants.

The model applies two methods for estimation of costs of the different abatement measures – partial equilibrium and engineering methods – which differ with respect to consideration of affected sectors' actual behaviour in the market. Partial equilibrium analysis is applied for calculations of costs of reductions in fertilisers, which rests on revealed behaviour on the fertiliser market. Data on prices and purchases of fertiliser can then be used for deriving costs of fertiliser reductions, which correspond to associated losses in profits. Market prices are also used for assessing costs of conversion of arable land into less leaching land uses such as wetlands and buffer strips. However, there is not enough data to evaluate the effect of massive land conversion on the market price of arable land, and the engineering method is therefore applied for cost calculations. Due to lack of data, the engineering approach is used for calculating costs for, not only land use changes, but also for all other abatement measures except for reductions in fertilisers.

Marginal cost of abatement measures in the drainage basins are calculated by combining estimated costs of cleaning measures with data on impact on the Baltic Sea which occurs by nutrient transports in air, soil and water. Measures affecting airborne emission have the most involved 'chain of impacts', where reductions in airborne emissions have direct and indirect impacts on the Sea. The direct impacts consist of the share of emission that would have been deposited on the Sea, and the indirect impacts occur through decreases in dispersal of deposition on land within the entire drainage basin, which, in turn, generates less leaching and final transport to the Baltic Sea. Measures with direct impact on the Sea, such as increased cleaning at sewage treatment plants located by the coast, have the most simple 'chain of impacts', where the impact on the Sea corresponds to the reduction at the source. Each abatement

measure is also subjected to capacity constraint, such as a maximum cleaning of phosphorus at sewage treatment plant by 90 per cent. Additional constraints consist of the number of households that can be connected to sewage treatment plants. Limitations on fertiliser and livestock reductions and land use changes are imposed in order to avoid drastic structural changes in the agricultural sector. For a detailed presentation of abatement capacities and costs of all measures see Gren *et al.* (2008).

References

Alkemade R., van Oorschot M., Miles L., Nellemann C., Bakkenes, M., ten Brink B., (2009). GLOBIO3: A framework to investigate options for reducing global terrestrial biodiversity loss. *Ecosystems*. DOI 10.1007/s10021-009-9229-5.

Braat, L. and ten Brink P., (2008). The cost of policy inaction. The case of not meeting the 2010 biodiversity target. Study for European Commission, DG Environment. Contract ENV.G.1/ETU/2007/0044.

http://ec.europa.eu/environment/nature/biodiversity/economics/teeb_en.htm

Carraro, C. (ed.) (2000). *Efficiency and equity in climate change policy*. Kluwer Academic Publisher, Amsterdam, the Netherlands.

Chadwick, D. R. 2005. Emissions of ammonia, nitrous oxide and methane from cattle manure heaps: effect of compaction and covering. *Atmospheric Environment*, 39, 787-799.

Clayton, H., I.P. McTaggart, J. Parker, L. Swan and K.A. Smith (1997). Nitrous oxide emissions from fertilised grassland: A 2-year study of the effects of N fertiliser form and environmental conditions. *Biology and Fertility of Soils*, 25, 252-260.

Dobbie, K.E. and K.A. Smith, 2003. Impact of different forms of N fertilizer on N₂O emission from intensive grassland. *Nutrient Cycling in Agroecosystems*, 67, 37-46.

Grasso, M. (2007). A normative ethical framework in climate change. *Climatic Change*, 81, 223-246.

Gren, I-M., Jonzon, Y., Lindqvist, M. (2008). Costs of nutrient reductions to the Baltic Sea – Technical report. Working paper 2008:1. Department of Economics, SLU, Uppsala.

Groenestein, C.M. and H.G. van Faassen. 1996. Volatilization of ammonia, nitrous oxide and nitric oxide in deep-litter systems for fattening pigs. *Journal of Agricultural Engineering Research*, 65, 269-274.

Helcom (2004). The fourth Baltic Sea pollution load compilation (PLC-4). Helsinki Commission. Helsinki, Finland.

Helcom (2007). An approach to set country-wise nutrient reduction allocations to reach good marine environment of the Baltic Sea. Helcom BSAP Eutro Expo/2007. Helsinki Commission, Helsinki, Finland.

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

<http://randd.defra.gov.uk/Default.aspx?Menu=Menu&Module=More&Location=Non e&Completed=0&ProjectID=15900>).

Holland, M., Watkiss, P., Pye, S., de Oliveira, A. and van Regemorter, D. (2005) Cost-benefit analysis of policy option scenarios for the Clean Air for Europe programme for Service Contract for carrying out cost-benefit analysis of air quality related issues, in particular in the clean air for Europe (CAFE) programme. http://www.cafe-cba.org/assets/thematic_strategy_analysis_v3.pdf

Jones, S.K., R.M. Rees, U.M. Skiba and B.C. Ball (2005). Greenhouse gas emissions from a managed grassland. *Global and Planetary Change*, 47, 201-211.

Jones, S.K., R.M. Rees, U.M. Skiba and B.C. Ball (2007). Influence of organic and mineral fertilizer on N₂O fluxes from a temperate grassland. *Agriculture, Ecosystems & Environment*, 121, 74-83.

Markowska, A. and T. Zylicz T. 1999. Costing an international public good. *Ecological Economics*, 30, 301-316.

Meijide, A. Diez, J. A. Sanchez-Martin, L. Lopez-Fernandez, S. Vallejo, A. (2007) Nitrogen oxide emissions from an irrigated maize crop amended with treated pig slurries and composts in a Mediterranean climate. *Agriculture, Ecosystems & Environment*, 121, 383-394.

MNP (2008). The Netherlands in a Sustainable World Poverty, Climate and Biodiversity. Second Sustainability Outlook. Netherlands Environmental Assessment Agency (PBL) publication number 500084003.

Oenema, O., G.L. Velthof and P.J. Kuikman (2001) Technical and policy aspects of strategies to decrease greenhouse gas emissions from agriculture. *Nutrient Cycling in Agroecosystems*, 60, 301-315.

Rabl, A., Spadaro, J. and van der Zwaan, B. (2005) *Environmental Science and Technology*, 39, 399-408.

Rosenthal, R. 2008. GAMS – A user's guide. GAMS Development Corporation, Washington, DC, USA

Sen, A. (1999). *Development as freedom*. Anchor, New York.

Smith, P. D. Martino, Z. Cai, D. Gwary, H. Janzen, P. Kumar, B. McCarl, S. Ogle, F. O'Mara, C. Rice, B. Scholes, O. Sirotenko, M. Howden, T. McAllister, G. Pan, V. Romanenkov, U. Schneider, S. Towprayoon, M. Wattenbach and J. Smith (2008). Greenhouse gas mitigation in agriculture. *Philosophical Transactions of the Royal Society B: Biological Sciences*, 363, 789-813.

Söderqvist, T. and L. Hasselström (2008). The economic value of ecosystem services provided by the Baltic Sea and Skagerrak., REPORT 5874 • DECEMBER 2008. <http://www.naturvardsverket.se/Documents/publikationer/978-91-620-5874-6.pdf>

Sommer S. G. and Petersen S.O. (1997) Nitrous oxide emissions from manure handling – effects of storage conditions and climate. In: Søren O. Petersen & Jørgen E. Olesen (ed.). Proceedings from an international workshop Helsingør, Denmark 24-25 January 2002. Danish Institute of Agricultural Sciences; Plant Production report no. 81.

Sommer, S. G. (2001) Effect of composting on nutrient loss and nitrogen availability of cattle deep litter. *European Journal of Agronomy*, 14, 123-133.

Sommer, S.G., S.O. Petersen, and H.B. Møller. 2004. Algorithms for calculating methane and nitrous oxide emission from manure management. *Nutrient Cycling in Agroecosystems*, 69, 143-154.

Tiedje, J.M. (1988). Ecology of denitrification and dissimilatory nitrate reduction to ammonium. In: Zehnder, A.J.B. (Ed). *Biology of Anaerobic Microorganisms*. New York, John Wiley and Sons, pp. 179-243.

Velthof, G.L., P.J. Kuikman & O. Oenema (2002) Nitrous oxide emission from soils amended with crop residues. *Nutrient Cycling in Agroecosystems*, 62, 249-261.

Velthof, G.L., O. Oenema, R. Postma & M.L. van Beusichem, 1997. Effects of type and amount of applied nitrogen fertilizer on nitrous oxide fluxes from intensively managed grassland. *Nutrient Cycling in Agroecosystems*, 46, 257-267.

Vinther, F.P., E.M. Hansen & J.E. Olesen, 2004. Effects of plant residues on crop performance, N mineralisation and microbial activity including field CO₂ and N₂O fluxes in unfertilised crop rotations. *Nutrient Cycling in Agroecosystems*, 70, 189-199.