Chapter 22

Costs and benefits of nitrogen in the environment

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Executive summary

Nature of the problem

- Single issue policies have been an effective means of reducing reactive nitrogen (N_r) emissions in the EU, but to make further reductions more-integrated approaches are required.

Approaches

- This chapter shows how cost–benefit analysis (CBA) can provide guidance for the setting of new policy priorities for the abatement of the European N_r emissions from an integrated perspective.
- Data on costs and benefits of N_r-abatement, including four national and regional case studies, are reviewed and made comparable by expression in euro per kg of added N_r (agriculture) or euro per kg of reduced N_r emission (unit cost approach).
- Social cost estimates are based on Willingness to Pay (WTP) for human life or health, for ecosystem services and greenhouse gas (GHG) emission reduction.

Key findings

- The total annual N_r-related damage in EU27 ranges between 70 and 320 billion Euro, equivalent to 150–750 euro/capita, of which about 75% is related to health damage and air pollution. This damage cost constitutes 1%–4% of the average European income.
- Inferred social costs of health impacts from NO_x are highest (10–30 euro per kg of pollutant-N_r emission). Health costs from secondary ammonium particles (2–20 euro/kg N), from GHG balance effects of N_2O (5–15 euro/kg N), from ecosystem impacts via N-runoff (5–20 euro/kg N) and by N-deposition (2–10 euro/kg N) are intermediate. Costs of health impacts from NO_3 in drinking water (0–4 euro/kg N) and by N_2O via stratospheric ozone depletion (1–3 euro/kg N) are estimated to be low.
- The first year social benefit of N_r for the farmer ranges between 1 and 3 euro per kg added N-fertilizer equivalent. Internalizing the environmental costs of N-fertilization would lower the optimal N-rate for arable production in North-West Europe by at least 50 kg/ha.

Uncertainties

- Major uncertainties in our approach are dose-response relationships and poor comparability of WTP studies. Also it is often not simple to identify the N_r-share in adverse impacts and in abatement measures.

Recommendations

- The CBA results presented provide support for the present focus of EU and UNECE N-policies on air pollution and human health, and on reducing ammonia emissions from agriculture; the social benefits of abatement tend to exceed the additional costs.
- Although options are attractive that offer simultaneous reductions of all N pollutants, the CBA points to the need to prioritize NO_x and NH_3 abatement over the abatement of N_2O emissions. Social cost of potential increases in emissions of N_2O and nitrate, when enforcing low ammonia emission techniques, are overwhelmed by the social benefits of decreased NH_3-emission.

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22.1 Introduction

Human welfare and nitrogen

In some parts of the world a considerable amount of anthropogenic reactive nitrogen (N_r) is lost to the air, water and land causing a range of environmental and human health problems while, simultaneously, in other parts of the world food production is N_r-deficient. Therefore, Galloway *et al.* (2008) conclude that 'optimizing the need for a key human resource while minimizing its negative consequences requires an integrated interdisciplinary approach and the development of strategies to decrease N-containing waste'. In fact the question is whether the N-cycle can be changed in such a way that a human welfare improvement is achieved, i.e. the benefits of this change exceed (or at least are in balance with) the associated costs. Given that many of the environmental effects involved, have costs and benefits for the population, government intervention might be required. Decisions by policy makers can therefore be supported by an integrated assessment of all social costs and benefits of changing N-management, showing the trade-offs that are at stake.

Human welfare implications of changing nitrogen management

Theoretically, the effect of a change in N-management on human welfare can be determined by comparing the social costs and benefits. The social costs are the resources a society has to give up when changing N management, e.g. the cost of an investment in sewage treatment or the income lost in agriculture if fertilizer use is limited. The social benefits are all effects that positively contribute to human welfare, e.g. the protection of threatened species or avoiding negative health impacts of NO_x emissions. A change in N-management will only lead to an improvement in human welfare if the sum of the social benefits exceeds the sum of the social costs. In theory, the social optimal level of N_r-use is found where marginal social cost equals marginal social benefits. In view of the complexity of the N cycle, in practice it is, however, a difficult task to determine all human welfare effects of changing N management, as will become clear in this chapter.

Policy context of social cost–benefit analyses related to nitrogen management

Increasingly, decision makers demand that decisions are based on an assessment of the social costs and benefits of N_r-reduction options. Two policy stages can be distinguished, requiring complementary assessment procedures.

(1) At the initial stage there is a relatively high level of environmental pollution, for which various low cost measures are available. If the pollution causes unacceptable social problems, the low cost of measures implies that further control will be beneficial for society. Therefore, in this stage policy makers are mainly interested in the identification of options that reduce pollution at the lowest costs and cost-effectiveness analysis (CEA) is a suitable tool to support policy making;

(2) At a later stage the environmental pollution has been reduced considerably through the implementation of low-cost measures and marginal benefits of further reduction have decreased. At this stage it becomes important (and often also more difficult) for policymakers to know more precisely the marginal cost and marginal benefits of N_r-reduction in order to determine economically efficient use levels and cost–benefit analysis is helpful to support policymaking.

In this chapter, Section 22.2 describes the theoretical background and main tools for integrated assessment of the social costs and benefits of N_r-reduction strategies. Section 22.3 gives examples and results of the economic valuation of the effects of N_r on (i) agriculture, (ii) ecosystems and (iii) human health. Section 22.4 addresses the cost of N_r abatement. Section 22.5 summarizes four relevant case studies of integrated assessment of costs and benefits. Section 22.6 provides some tentative CBA results from EU27 countries and the agricultural sector and discusses the limitations and potential of CBA in developing N_r-reduction policies.

22.2 Theoretical background from an economic perspective

22.2.1 General framework

An economic impact assessment of policy measures to prevent and mitigate the harmful effects resulting from N_r-use requires assessment of all the welfare effects of these measures. Various types of policy measures with different impacts on social welfare are available. Figure 22.1 shows how various response options can influence the environmental impact of economic activities (agricultural production, energy use, and industrial processes) that serve to meet the demand for various goods and services. Specific policies might either change the driving forces behind the N_r-related environmental impact, i.e. the demand for goods and services, affect the economic activities (leading to reduction or prevention of emissions to the environment), work towards restoring the environmental quality, or deal with the final impacts, e.g. medical treatment or ecosystem restoration. These options have an impact on welfare in several ways. In many cases it is, however, far from a simple matter to quantify and value these welfare effects in a directly comparable way.

22.2.2 Methods for economic impact assessment

Important economic impact assessment methods include cost-effectiveness analysis (CEA) and cost–benefit analysis (CBA). The purpose of CEA is to find out how predetermined targets, e.g. threshold values for nutrients in a catchment, can be achieved in the least cost way (Brouwer and De Blois, 2008). In practice, CEA is carried out at varying levels of complexity, scale, comprehensiveness and completeness. The same applies to CBA, which can be carried out from the perspective of an investment decision of an individual company, accounting for the private costs and revenues only, but also from the perspective of society as a whole taking into account all effects that influence social welfare. The latter is also referred to as 'extended'

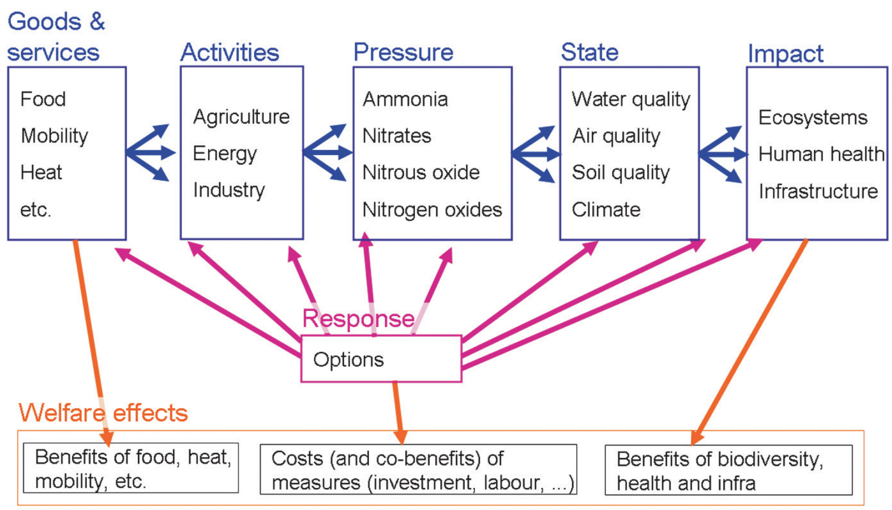

Figure 22.1 Schematic representation of general framework.

CBA (Brouwer and van Ek, 2004). We refer to CBA here primarily from a social perspective, in which case CBA compares the costs and benefits of different policy options, preferably in monetary terms. When costs and benefits of policy options occur at different points in time, they are made comparable in time through a weighting procedure called discounting (see the later section on Discounting in Section 22.2.5). The result of this analysis is a net present value, where the discounted (present) values of the costs are subtracted from the discounted (present) value of the benefits. Dividing the present value of the benefits by the present value of the costs provides a B/C ratio, which, if larger than one, indicates that the policy option is beneficial. Monetizing the impacts of public environmental policy is not always possible, however, non-monetized impacts, if considered relevant, can be included qualitatively during the discussion of the CBA results. Different approaches exist on how non-monetized impacts are included in a CBA: they can be listed as 'pro memoria' items in the balance sheet, expressed in qualitative or quantitative terms or become part of a wider multi-criteria analysis (Brouwer and van Ek, 2004).

22.2.3 Valuing the benefits

The monetary value of a positive welfare effect, e.g. an improvement in water quality, can be measured as the amount of money society is willing to give up to secure the improvement (De Zeeuw *et al.*, 2008). This is called the willingness to pay (WTP). For goods and services that are traded in markets, this WTP can be derived from demand for the goods and services involved at different market prices. However, for many environmental goods and services there exist no markets and therefore no market prices are available which reflect their economic value. Therefore, alternative ways are required to estimate the monetary value of changes in environmental quality. Although it seems that the acceptability of these methods has increased due to substantial improvements in the state-of-the-art of the methodological approach, there remains discussion about whether or not we should always try to put a monetary value on all environmental goods and services (De Zeeuw *et al.*, 2008). It is perhaps also important to point out that CBA is a tool to support decision-making. Policy makers are not bound to follow the outcome of the CBA, they may apply their own weightings to the wide variety of welfare implications of public environmental policy or factor in other issues that lead them to a different conclusion than that generated by the CBA.

22.2.4 Calculating the cost

Costs are defined as the value of the negative welfare implications of an activity or policy, resulting in the sacrifice of alternatively employable scarce resources (Markandya *et al.*, 2001). Relevant costs are not only the costs that a typical operator or farmer faces when implementing an abatement measure, but include all resources a society has to sacrifice as a consequence of the N management practices. These costs include investment and operating cost (e.g. in the case of an investment in sewage treatment), but also opportunity cost, for example by loss of productivity of agricultural land when it is set aside to reduce N_r leaching, or welfare loss because of limitations to recreational activities in a certain area as a result of a measure.

Like the assessment of benefits, the assessment of costs faces several challenges.

- The potential for the actual response to policies tends to differ from the predicted response. For example, industries will always look for the cheapest solution to meet legislated requirements and experience shows that these may not always match the technical solutions typically included in cost assessments (see e.g. Oosterhuis, 2007).
- Actually incurred costs of measures tend to be lower than predicted, owing to technological improvements, added competition and the fact that stakeholders who perceive themselves most likely to be disadvantaged by new policy are those most likely to respond to surveys on cost data (see e.g. Oosterhuis, 2006).
- Cost estimates tend to focus on single pollutants or even single environmental issues (e.g. eutrophication), while substantial cost savings may be achieved when assessing the co-benefits.

Notwithstanding the above, the expenditures (and also earnings or cost savings) directly associated with the implementation of the policy option by the relevant actors (e.g. the farmers or the government) can generally be estimated. From the perspective of social welfare, costs (and savings) for all parties within a society have to be considered (Brouwer *et al.*, 2008). These include the indirect costs, i.e. costs for other actors than those implementing the measure (e.g. a reduction in pig production in a given area would imply lower economic activity among people providing services and products to these farms and for those who buy their products from these farms). In practice, these costs are more difficult to estimate and require broader macro-economic models. The relevance of including the indirect costs depends on their expected size and the role they are expected to play in the decision-making procedure (Zhang and Folmer, 1998).

22.2.5 Use of CBA in policymaking

Potential and conditions

CBA has an important role in the development, design and evaluation of policies that influence the N cycle. Its central role in informing policy decisions is mainly due to its ability to develop consistent optimal policies, provide accountability to decision makers and answer questions regarding the potential alternatives.

The first key benefit of CBA is that it fosters the development of a robust evaluation methodology that can be applied across a range of policy decisions. This will help to avoid inconsistencies of the type where a million is spent in one area to avoid the loss of, e.g., a life year while refusing to spend a thousand for the same in another area. A consistent approach also provides a key mechanism to allow scrutiny of policies and determine the accountability of decision makers. By producing an auditable and open CBA it demonstrates in detail to stakeholders exactly the reasoning behind the decision. This explanation allows all stakeholders to review the methodology that has been employed, the assumptions that were made and the sources of information. It should also be noted that such an approach does not preclude the influence of impacts that could not be reflected in the CBA, but does highlight these factors for scrutiny.

Scale

The scale at which CBA will be performed differs between different problems, because of differences of the dose–response functions and of the social appreciation of the costs and benefits. A CBA for N effects and measures in urban air pollution will be very different from a CBA for marine eutrophication (see e.g. Jacobsen *et al.*, 2007). Therefore, when applying CBA for the complete N-cycle, breaking down the problem to a smaller scale is necessary.

Dealing with uncertainties

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: data uncertainties which can often be described using statistical distributions and dealt with accordingly; modelling assumptions that may be approached using sensitivity analysis; and 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 minimize costs once a policy is in force, the tendency is for this source of bias to exaggerate costs. For a discussion of a few ways to deal with various kinds of uncertainties in the context of using CBA in decision-making, see Supplementary materials, referenced at the end of the chapter.

Distributional impacts

As an illustration of the relevance of distributional impacts of policy options, consider a policy that increases the wealth of 10% of the population with the highest wealth by 100 million euro at the cost of 95 million euro from the 10% of the population with the lowest wealth. Taken as an aggregate the population is 5 million euro better off financially. However, that does not necessarily mean that total social welfare has increased, and in this example it may even have been reduced. To partially reflect such considerations, some Member States, such as the UK, have produced distributional weights by income (Treasury, 2010). A second aspect relates to the population over which the impacts are dispersed. This could become a policy consideration, particularly if the costs are borne by a relatively small group, whereas the benefits are distributed over a much larger population.

In addition to the specific work that has been done to try and reflect such equity considerations, there is generally also a requirement to reflect the distributional impacts. Such requirements are provided in impact assessment guidance both at the EU level and in many member states such as the UK.

Discounting

Discounting is used to express time-preference and make costs and benefits that occur at different points in time comparable. It is obvious that the choice of discount rate is important for the outcome of a CBA. This is in particular the case if costs and benefits occur at very different points in time, which is often true for policy options with respect to environmental quality (e.g. climate change, see Markandya *et al.*, 2001). Even for discount rates that are not excessively high, it follows that costs and benefits to future generations are practically ignored (De Zeeuw *et al.*, 2008).

Side effects

Measures that are aimed at reaching an environmental objective, e.g. good ecological status for rivers, will often have additional effects on the environment, which might either positively or negatively affect social welfare. In the literature, several terms are used to depict the associated benefits and costs that arise in conjunction with mitigation policies for a specific purpose, including co-benefits, ancillary impacts, secondary benefits and side effects (Markandya *et al.*, 2001). Examples of side effects related to agricultural measures to reduce ammonia emissions are a change in greenhouse gas emissions (Brink *et al.*, 2005) and reductions in odour from animal production. The cost effectiveness analysis can in some cases be more complex when both multiple primary and secondary effects are involved as well as upstream–downstream and transboundary effects are observed (Jacobsen, 2007). Although from a policy perspective policy options have primary and secondary effects, from a social welfare perspective all effects are relevant. Therefore, it is obvious that in a social CBA all (intended and unintended) effects have to be considered.

22.3 Valuation of nitrogen effects

22.3.1 Introduction

Reactive N is beneficial for society as it is a key component of chlorophyll, amino acids, proteins and enzymes (see e.g. Olson and Kurtz, 1982). Sufficient supply of N_r is required for plant metabolism, and addition of N_r will essentially increase the efficiency of photosynthesis to produce carbohydrates for food, feed, fibre, etc. In view of the relatively low price of N fertilizer as compared to the value of land, labour and crops, application of N_r is beneficial for farm economy up to high rates (see also Jensen *et al.*, 2011, Chapter 3, this volume). These high rates of artificial fertilizer in combination with inefficient handling and use of manures are the reasons that agriculture is now the dominant source of emissions of N_r to the environment in many parts of Europe (Leip *et al.*, 2011, Chapter 16 this volume). The other major source of N_r is energy use, where formation and emission of nitrogen oxides is a side-effect of combustion of fossil fuels. These emissions cause social cost through impacts on ecosystems, human health and the GHG-balance.

As explained in Section 22.1, the optimal level of N-mitigation for society is reached when the marginal cost of mitigation is equal to the marginal benefit of reduced environmental impacts. In the case of agriculture, social damage costs can be mitigated by reducing the N_r-input. The marginal cost of these reductions equal the benefits lost due to decreased crop yield. Von Blottnitz *et al.* (2006) determine the so-called socially and privately optimal rates of N for agricultural production; SONR and PONR (See Box 22.1). Mitigation options other than reducing the N_r-rates are not considered.

The benefits of N_r for agriculture can relate both to mass and quality of the crop. Crop mass typically shows a non-linear response to the N_r input rate, with diminishing, or for some crops negative, return with increasing rates. The three major damage categories for society are (i) loss of life years and human health, (ii) loss of biodiversity and ecosystem services, (iii) climate change. As these social impacts have multiple causes and strongly depend on resilience and resistance of, respectively, humans, ecosystems and climate, major problems exist to derive causal relations with emissions of N_r, and to value the N_r-share of the impact. An overview of unit benefits (P_C) and damage costs (E) of N_r follows.

Box 22.1 Equations for CBA of nitrogen in agriculture

$(dY/dN)_{PONR} = P_N/P_C$

$(dY/dN)_{SONR} = (P_N + E)/P_C$

where:

Y = crop yield (kg/ha)

N = input rate of reactive nitrogen (kg/ha)

PONR = privately optimal N_r input rate (kg/ha)

SONR = socially optimal N_r input rate (kg/ha)

P_N = price of N_r (purchase and handling; euro/kg)

P_C = price of crop (euro/kg)

E = externalities; sum of environmental damage costs (euro/kg)

$E = \Sigma D(Ni)\, P_D$

where:

D = social damage caused by nitrogen (euro/impact)

Ni = emission of N_r compound (kg/ha)

Ni = ef A or ef N

ef = emission factor

A = economic activity

P_D = social cost of environmental damage (impact/kg N_r)

UBoN = $(Y_{PONR} - Y_{N=0})*P_C/\text{PONR} - P_N$

where:

UBoN = net unit crop benefit of N_r (euro/kg N)

22.3.2 Benefits for agriculture

The economic benefit of N_r for the farmer depends on prices of crops and fertilizer. Although crop prices are somewhat volatile, increasing prices of artificial fertilizer (Jensen *et al.*, 2011, Chapter 3 this volume) cause the price ratio P_N/P_C to increase in time. As a result, the marginal net benefit of N_r for farm economy also tends to decrease. Extrapolation of the trend of the price ratio between 1995 and 2008 to 2020 would give a price ratio of 10 as compared to present values of 7. Values of PONR in 2020 would then be about 15 kg/ha lower than present values. On the other hand uncertainties about the (non-linear) response of crop yield to N_r-rate cause unit benefits of N_r (tangent in Figure 22.2) to be uncertain and consequently also the value of PONR. From the perspective of the farmer as a risk manager, this uncertainty of response in combination with uncertainty about weather conditions during the oncoming growing season (in fact there is a suite of possible response curves), may cause him to focus more on the average (chord in Figure 22.2) than on marginal (tangent in Figure 22.2) economic return on his investment in N_r. This behaviour of the farmer is amplified by the small share of costs of N_r in the total variable production costs. Pedersen *et al.* (2005) showed that the N-costs for potato farming in six out of seven

Table 22.1 Unit benefits of reactive N in euro per kg of N for various crops on sandy soils as a function of reduction of N-fertilizer use relative to 2006 (in general the fertilizer recommendation) in the Netherlands (values are gross benefits as savings on fertilizer, or loss of income from manure acceptance are not considered; inferred from van Dijk *et al.*, 2007)

	N-fertilization relative to 2006 (= recommended) levels				
	100%–90%	90%–80%	80%–70%	70%–60%	60%–50%
	crop benefits of N (euro per kg N_r)				
Edible potato	2.1	2.6	3.0	3.8	4.2
Starch potato	0.3	1.0	1.7	2.3	2.7
Sugar beet	0.3	1.7	2.7	4.0	5.0
Silage maize	0.5	0.8	1.4	1.4	1.6
Leek	2.0	3.3	4.3	5.5	6.9
Broccoli	7.0	9.4	11.9	14.6	17.4
Cauliflower	3.3	5.4	7.4	9.5	11.9
Tulip bulb	6.3	10.0	13.5	18.0	22.3
Lily bulb	13.2	19.4	26.1	33.2	31.3

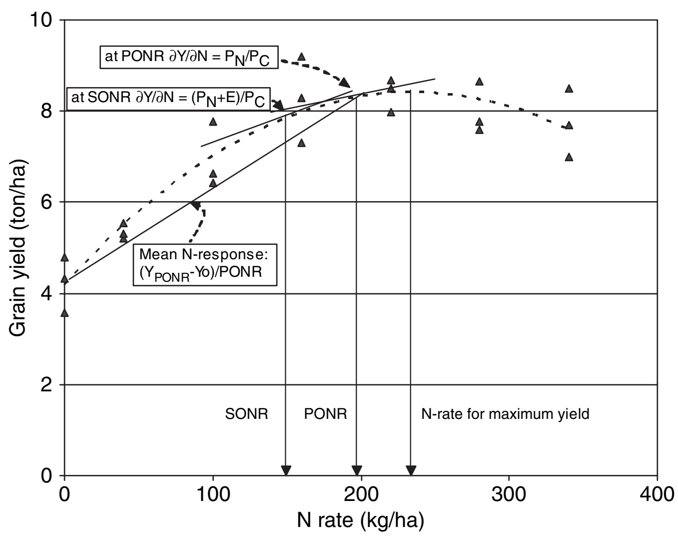

Figure 22.2 Example of a yield response curve for winter wheat in the UK demonstrating the marginal and average unit benefit of annual nitrogen fertilizer inputs.

European Member States did not exceed 5% of variable costs. N-cost for cereal production, however can amount to 20%.

Effect of crop type and N-level

Using yield curves based on field trials, commonly used second order polynomial fits and 2006 price levels (Van Dijk *et al.*, 2007) indicative unit crop benefits (euro) per kg of N_r (UBoN) can be derived for a range of arable and horticulture crops (Table 22.1).

Within a range of 0% to 20% below the recommended N fertilization level, unit benefits range between 0.5 and 20 euro per kg N_r (Table 22.1) and unit benefits will increase when N-fertilization is further reduced. Unit benefits for vegetables and flowers are clearly higher than those for arable crops and return on investment in N-fertilizer for these crops is certain. Dividing UBoN by the price of fertilizer-N yields the net financial return on the investment in fertilizer (euro per euro). At fertilizer prices in 2006 of 0.8 euro per kg N_r (calcium ammonium nitrate, CAN) N-fertilizer levels in 2006 were exceeding PONR for starch potato, sugar beet and silage maize. In addition to the tendency for the N-recommendation to exceed PONR, farmers tend to apply more N-fertilizer than recommended.

Unit benefits of N for winter wheat, oilseed rape and dairy in Europe

Winter wheat is the major arable crop in Europe using about 25% of agricultural area and total N-fertilizer use in Europe (EU27), and was therefore selected to provide some more information on variation and uncertainty of N_r benefits. The data used represent a wide range of conditions, rotations and setups of field trials. The net unit benefit of N_r (UBoN: $(Y_{opt} - Y_o)*P_C/\text{PONR} - P_N$) ranges from 0.4 euro per kg N_r in SE Europe to 2.7 euro per kg N_r in NW Europe. Oilseed rape is an emerging crop but the market and price for it is uncertain because of current developments in EU policies for climate and biofuels. Values of UBoN for oilseed rape (Figure 22.3) range between 0.3 and 2.3 euro per kg and therefore are somewhat lower than for winter wheat in view of lower yield and weaker response to N_r.

The share of roughage and feed concentrates use for the dairy sector in Europe varies strongly; with high shares of roughage in regions with the largest areas of productive grassland and low land prices (northern 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 per kg N_r (Figure 22.3) and therefore similar to values for wheat in NW European countries. UBoN values for production

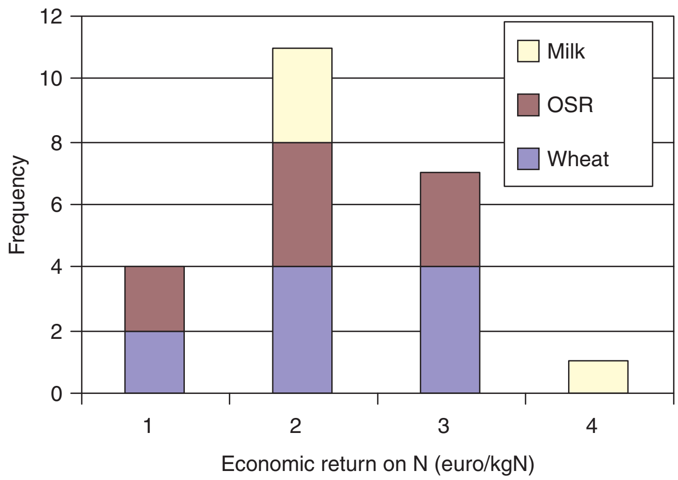

Figure 22.3 Economic return in first year an investment on nitrogen fertilizer for grain of wheat and oil seed rape (OSR), and for milk, using results of field trials ($N = 23$) and based on a wheat price of 125 euro/ton, a OSR price of 220 euro/ton and a milk price of 290 euro/ton and a CAN-fertilizer price of 0.8 euro/kgN.

of roughage for silage for these four cases range between 0.6 and 2 euro per kg N_r using a herbage price of 100 euro per tonne.

Combining UBoN results for 23 data sets from field trials for winter wheat, oilseed rape and dairy (Figure 22.3) a median value of UBoN of 1.7 euro per kg N_r was found (see Supplementary materials).

There is evidence that in the EU protein supply to livestock is in surplus. Witzke and Oenema (2007) estimated protein surplus to range between 10% and 20%. Furthermore, statistical data on N excretion of livestock show substantial variation within one member state (e.g. for fattening pigs in the Netherlands about +/– 20%) indicating there is room for efficiency gains. The implementation of low-protein animal feeding is not without cost for the farmer as this will restrict the use of certain types of inexpensive industrial residues and raw materials in the compound feed industry. Oenema *et al.* (2009) estimated the total cost of a 10% reduction of protein in feed at 10 billion euro/yr for EU27.

Uncertainties in the net unit crop benefit of N (UBoN)

Values of UBoN are uncertain and change in time, which are uncertainties a farmer has to deal with when deciding on the proper N_r-rate for his crops. Uncertainties include the following.

- The shape of yield response curve: the value of PONR, and consequently also the value of UBoN, is sensitive to the shape of the response function. Gandorfer (2006) and Henke *et al.* (2007) used different equations to fit yield response of winter wheat and found a PONR range of +/– 15%. The corresponding range of UBoN is about 1 euro per kg N_r.
- Annual variation of weather: PONR for winter wheat for individual years ranged between 145 and 235 kg per ha; the resulting uncertainty of UBoN is about 3 euro per kg N_r.
- Use of N in manure: about one third of N-supply in agriculture in EU27 comes from manure-N (Velthof *et al.*, 2009). However, effective supply is much less, as it is economically unattractive, both for cattle and arable farmers to use manure-N, in view of costs of transport, application and uncertain composition and lower nutrient efficiency. It is not uncommon that in areas with high livestock densities manure N has a negative economic value (and UBoN).
- Prices of food and N-fertilizer: when farmers fertilize at the economically optimal level, UBoN is relatively insensitive to the price ratio.

Unit benefits of N for crop production based on trials are underestimates, as in these trials the yield at zero N-addition is on average half (35–80% for the 23 trials in Figure 22.3) of the maximum yield. So to a large extent crops grow on N-resources from the soil and previous crops. For the case studies used for Figure 22.3, estimates of these N-resources are in the range of 70–200 kg per ha and clearly related to N_r-fertilization prior to the trial year, and to some extent on other N_r-inputs like biological fixation of N_2 and atmospheric N_r deposition. Atmospheric N_r deposition on agricultural soils in the EU is about 3 Mt per yr and constitutes about 20% of the total input (Leip *et al.*, 2011, Chapter 16 this volume) and a much larger share for organic farming. Including long-term effects of N_r-fertilization on crops yield would on average double the unit benefit. Using data from (rare) long term field trials, like the Broadbalk Experiment at Rothamsted (Brentrup *et al.*, 2004) for continuous winter wheat, gave a UBoN value of 3.5 euro per kg N_r and therefore 1–2 euro per kg higher than for annual trials.

22.3.3 Impacts on ecosystem health and function

Valuation of the impacts on biodiversity

A useful starting point for economic valuation of impacts of N on ecosystems is the Millennium Ecosystem Assessment's classification of ecosystem services (MEA, 2005). This classification focuses on four ecosystem service (ESS) types provided by the natural environment that have positive or negative consequences for human populations: (i) provisioning services, (ii) regulating services, (iii) supporting services and (iv) cultural services (Figure 22.4). Following this framework, biodiversity is therefore valued through its impact on the different ecosystems' ability to provide key services underpinning human well-being.

Although, biodiversity valuation is inherently an interdisciplinary area of study, it is clear from review studies (Nunes *et al.*, 2003; Markandya *et al.*, 2008; Raffaelli *et al.*, 2009) that existing studies are either dominated by a social science perspective (O'Neill, 1997; Lee and Mjelde, 2007) or a natural science perspective (Costanza, 1980; Odum and Odum, 2000).

Valuation of loss of ecosystem services due to N

Although the TEEB-COPI study (Braat and ten Brink, 2008) did not make the role of N_r explicit, it lists (only) three studies that provide data to derive unit cost values for ESS linked to N. A value of 2.2 euro per kg N_r was given for the ESS 'Water purification and waste management' both for scrubland and grassland, and 25 euro per kg NO_x-N for the ESS 'Air quality maintenance'.

Pretty *et al.* (2003) quantified costs of freshwater eutrophication in England and Wales. The problem with using this

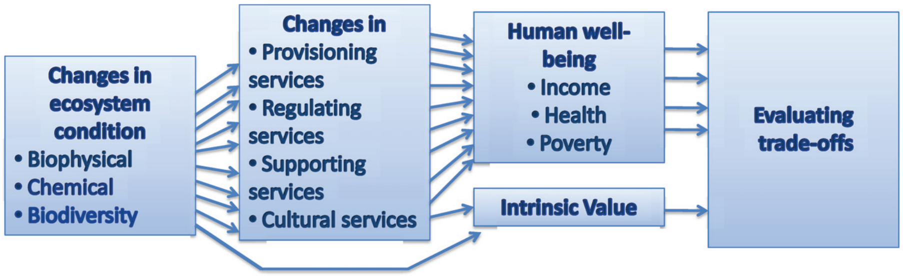

Figure 22.4 Mapping the link between biodiversity, ecosystem services and human well being (source MEA, 2005).

study for deriving unit costs is twofold: (1) no distinction is made between the effect of N_r and phosphorus, and (2) damage costs have a mixed background and some cases are in fact control costs. Cost items considered are reduced value of waterside properties, drinking water production, reduced recreational and amenity value, and also loss of biodiversity. They estimate total damage cost due to loss of ESS at 80–120 million euro per year (105–160 US$). Considering a total N_r-runoff (waste water and agricultural runoff) in the UK of around 300 kt per year in 2000 and attributing all loss to N, a unit cost of 0.3 euro per kg can be inferred.

Söderqvist and Hasselström (2008) estimated WTP for a clean Baltic (see also Section 22.5.2) updating results from Contingency Valuation surveys in the 1990s. In this survey a random sample of respondents was questioned in the 1990s about their Willingness to Pay (WTP) for a Baltic Sea 'undisturbed by excessive inputs of nutrients'. The causality and share of N for eutrophication of the Baltic Sea was not made explicit, but instead the WTP for the Baltic Sea objective was made equivalent to a reduction of 50% of the N_r-load. Values of WTP range between 70–160 euro per household for the Eastern European Baltic states with lower GDP, and between 500–800 euro per household WTP values for the Baltic States with high GDP (Table 22.2). Values are somewhat higher than values reported in the AQUAMONEY study for 11 river basins that ranged between 20–200 euro per household (AQUAMONEY, 2010).

Assuming that eutrophication damage can be mitigated by a 50% reduction of the N_r-load to the Baltic Sea, WTP results can be converted to an average unit damage cost of 12 euro per kg N_r for the total Baltic drainage basin, and range between 2–6 euro for East European Baltic states and 23–42 euro for the NW European Baltic states (Table 22.3). Gren *et al.* (2008) report a range of unit damage costs of 12–24 euro per kg N_r, based on Söderqvist and Hasselström (2008), using different discount rates.

The NEEDS project (New Energy Externalities Developments for Sustainability; Econcept; Ott *et al.*, 2006) is one of the few studies that has attempted to estimate the value of the loss of biodiversity due to acidification and eutrophication across European countries. The authors state that they use a restoration cost approach implicitly assuming that society is willing to bare the costs of restoration and that the cost estimates therefore offer a lower bound estimate of the benefits involved with restoration. Typical results from NEEDS are costs to restore occurrence of target species that have disappeared due to atmospheric deposition of eutrophying and acidifying N_r compounds. These values were converted to provide (low) estimate of average unit damage cost for EU25 of 2.5 euro per kg for NO_x-N (range 0.4–10) and 2.3 euro per kg NH_3-N (range 0.1–10) (for further details see Supplementary materials).

22.3.4 Impacts on human health

There are several routes by which N pollutants can affect human health leading to a variety of impacts (Table 22.4. see also Townsend *et al.*, 2003).

In the following paragraphs, dose–response relations and economic value are discussed for all listed impacts, except, due to lack of information, for odour and global warming.

Air pollution

For air pollutants, NO_x is a precursor of O_3 which is harmful to human health (Moldanová *et al.*, 2011, Chapter 18 this volume). The evidence for direct effects of NO_2 is less clear and most health impact assessments (including CAFE; Holland *et al.* 2005a,b) have not assumed direct effects; instead they evaluate the health damage of NO_2 by applying the dose–response functions of ambient PM to the nitrates that are created in the atmosphere from NO_x emissions (health impacts from secondary particulate matter are highly uncertain and debated; for more detail, see Moldanová *et al.*, 2011).

The monetary value includes market costs (medical treatment, wage and productivity losses, etc.), as well as non-market costs that take into account an individual's Willingness-to-Pay (WTP) to avoid the risk of pain and suffering. If the WTP for a non-market good has been determined correctly, it is like a price, consistent with prices paid for market goods. The range of mean annual health cost in EU Member States is 2–32 euro per kg N for NO_x, and 2–36 euro per kg N for NH_3 (Table 22.5).

The most important endpoint for air pollution by N_r is mortality from chronic exposure (to ozone and secondary particulate matter) which contributes 67% to health cost. As shown by Rabl (2003), air pollution mortality must be evaluated in terms of loss of life expectancy rather than of number of premature deaths. Thus, one needs the value of a life year (VOLY). But, by contrast to the numerous studies of so called 'Value of Statistical Life' (VSL; an unfortunate and often misunderstood name for what is really the 'willingness to pay for avoiding the risk of an anonymous premature death'), there have been very few studies until now to determine VOLY. For the 1998 and 2000 reports ExternE had calculated VOLY by assuming that VSL is a discounted sum of annual VOLYs; choosing 3.4 million euro

Table 22.2 Benefits of accomplishing a 'clean' Baltic Sea, converted to willingness to pay (WTP) per household of four people and to N damage costs for ecosystem aspects assuming that by a 50% reduction of the N_r-load a 'clean' Baltic will be accomplished

	WTP benefits Euro ×10^6	N_r-Load Gg	Population[a] Million 2002	WTP euro/ household	Damage cost Euro per kg N_r
Denmark	920	44	4.5	823	42
Estonia	60	56	1.4	168	2
Finland	610	49	5.1	475	25
Germany	530	46	2.8	746	23
Latvia	60	44	2.4	102	3
Poland	930	318	38.6	96	6
Sweden	1460	74	8.8	664	39
Russia	180	83	9.7	74	4
Lithuania	80	93	3.6	89	2
Total	4830	807	76.9	251	12

Note: [a] In the area draining to the Baltic; data from Hannerz and Destouni, 2006.

Table 22.3 Summary of unit cost estimates for ecosystem damage by reactive nitrogen

Study	Parameter	Valuation technique	Unit cost (euro per kg N_r)
Eutrophication in UK (Pretty *et al.*, 2003)	Loss of biodiversity and services	Mixed	0.3
Eutrophication Baltic (Gren *et al.*, 2008)	Clean un-eutrophied Baltic	WTP	12–24
Modelling impacts of atmospheric deposition on terrestrial systems NEEDS (2006)	Disappeared fraction of target species	Ecosystem restoration	3 (NO_x) 2 (NH_3)
TEEB/COPI (EC, 2008)	Loss of ecosystem services	WTP ecosystem services	2–25

for VSL (a weighted mean of European studies), this implied a VOLY of approximately 100 000 euro/life year. The Current ExternE recommendation for VOLY is 40 000 euro per life year, based on a contingent valuation by the ExternE team in nine countries of Europe with a total sample size of almost 1500 (Desaigues *et al.*, 2007). The European commission currently recommends a range of 52 000 to 1 20 000 euro per life year.

Nitrate in drinking water

The threat to human health of nitrate in drinking water was described in an earlier chapter (Grizzetti *et al.*, 2011, Chapter 17 this volume). Although a regulatory limit of 50 mg/l for nitrate in drinking water has been in place in the EU since 1980, there are no reliable dose–response functions to assess health loss or mortality due to nitrate in drinking water. Epidemiological studies providing evidence for health impacts are rare: the European Food Safety Authority (2008) concluded from a review of recent epidemiological studies that even 'these were mostly studies with a weak study design and limited strength of evidence'. While there is consensus that the association between nitrate and methaemoglobinaemia is weak, there is emerging evidence for increased incidence of colon cancer to be one of the more prominent chronic health impacts of nitrate in drinking water exceeding 25 mg/l nitrate (DeRoos *et al.*, 2003; Van Grinsven *et al.*, 2010; Grizzetti *et al.*, 2011, Chapter 17, this volume). Using data for 11 EU member states the total population exposed to drinking water exceeding 25 mg/l nitrate was estimated at 23 million persons (6.5% of the total population) of which 8 million persons (2.3%) were exposed through public supply (Table 22.6). The associated increase of incidence of colon cancer was estimated at 3% (Van Grinsven *et al.*, 2010).

The total monetary value of this loss of life was estimated at 1.6 billion euro per year or 4.5 euro per average individual, using a value of 40 000 euro/yr for years lost due to premature death and 12 000 for years of health lost due to suffering from colon cancer.

The mean unit damage cost for the 11 member states is estimated at 0.7 euro per kg of N-leaching (range 0.1–2.4 euro per kg N_r) when assuming that a 100% reduction of N-leaching is required to fully prevent exceedance of 25 mg/l NO_3. The lowest

Table 22.4 Overview of N_r related health impacts

Pollutant	Health impacts and routes	Health impacts	Unit damage cost
NO_x	Inhalation - impacts via O_3 - impacts via PM_{10} - direct impacts of NO_2	Asthma, respiratory disorder, inflammation of airways, reduced lung functions, bronchitis, cancers	5.6 euro/kg NO_x (euro price level 2000 inferred from CAFE results)
NH_3	Inhalation: - direct impacts (negligible) - impacts via PM_{10} - odour	See NO_x Small as odour contribution by NH_3 is modest	9.5 euro/kg NH_3 (euro price level 2000 inferred from CAFE results)
N_2O	Health impacts from global warming, often enhanced by eutrophication	Enhancement of vectors for infectious diseases (malaria) and frequency of infestations (HABS, insects)	
Nitrate	Drinking water intake followed by conversion to nitrite. Nitrite is a precursor for carcinogenic N-nitroso compounds and nitrite binds to haemoglobin	Cancers (e.g. colon, neural tube) and reproductive outcome from chronic exposure. Methaemoglobinaemia (blue baby disease)	0.7 euro/kg NO_3-N For exposure through public and private wells using groundwater

Table 22.5 Unit damage costs for health impacts by airborne NO_x and NH_3 (euro per kg N_r; using VOLY 40 000 euro per life year and the CAFE/WHO methodology (Methodex, 2010)

	NH_3 euro per kg N	NO_x euro per kg N		NH_3 euro per kg N	NO_x euro per kg N
Austria	15	29	Latvia	4	5
Belgium	36	17	Lithuania	2	6
Czech Republic	24	24	Luxembourg	30	29
Denmark	10	14	Netherlands	27	22
Estonia	3	3	Poland	12	13
Finland	3	2	Portugal	4	4
France	15	25	Slovakia	17	17
Germany	22	32	Slovenia	16	22
Greece	4	3	Spain	5	9
Hungary	13	18	Sweden	7	7
Ireland	3	12	United Kingdom	21	13
Italy	13	19			

values are found for Ireland, the UK and the Netherlands and could be viewed as benefits of investments in a good drinking water infrastructure and protection against nitrate pollution. Highest values are found for Austria, Denmark, Italy, France and Germany and in part reflect high nitrate leaching rates and a lower proportion of the population connected to large public drinking water supply facilities.

The unit damage cost results are tentative. In view of ongoing discussions on clinical and epidemiological evidence for adverse health of nitrate, a lower bound for the unit damage of zero seems appropriate. Unit damage cost could also be higher, for example, when other chronic health impacts are included, or by including drinking water from surface water resources and data from central and eastern Europe or when present exceedance of 25 mg/l NO_3 is attributed to a smaller (e.g. 50%) share of present N leaching.

Depletion of stratospheric ozone by N_2O

Depletion of the stratospheric ozone layer by man-made chemicals causes skin cancers and cataract. Owing to the large reduction of emissions, such as hydrochlorofluorocarbons and halons, after implementation of the Montreal Protocol, nitrous

Table 22.6 Assessment of increased incidence and damage costs for nitrate in drinking water derived from groundwater resources, in 11 EU member states using groundwater and drinking water quality data, and cancer registration data for the mid 1900s (Van Grinsven *et al.*, 2010)

	Total population exposed to >25 mg/l NO_3	Colon cancer incidence (1993–1997)	Additional colon cancer cases due to nitrate	Total number of lost healthy life years before death	Total number of lost life years from premature death	Monetary values of loss of (healthy) life years	Monetary values of loss of (healthy) life years	Unit health damage cost from N-leaching agricultural land
	million	×1000	×1000	×1000	×1000	million euro/year	euro/capita	euro/kg N
Austria	8.8	2.5	0.1	0.5	0.4	23	2.9	1.9
Belgium	5.9	3.7	0.1	0.5	0.4	23	2.2	2.4
Denmark	16.2	2.0	0.2	0.8	0.6	35	6.6	0.6
Finland	3.4	1.2	0.0	0.1	0.1	4	0.9	0.8
France	9.7	19.8	1.0	4.7	3.6	202	3.4	0.6
Germany	8.9	42.0	1.9	9.1	7.1	393	4.8	1.4
Ireland	2.8	1.1	0.0	0.1	0.1	3	0.9	0.1
Italy	6.8	28.3	1.0	4.7	3.6	202	3.5	1.9
Netherlands	3.5	5.5	0.1	0.5	0.4	20	1.3	0.2
Spain	3.8	12.6	0.2	1.2	0.9	51	1.3	0.4
UK	2.0	20.4	0.2	1.0	0.8	43	0.7	0.2
Total		139.2	5	23	18	1000	2.9	0.7

oxide presently is the dominant ozone depleting substance (ODS; Ravishankara *et al.*, 2009). Total health loss by ozone depletion in 2007 was estimated at about 500 000 disability-adjusted life years (DALY) by Struijs *et al.* (2010).

A first estimate of the unit damage cost for human health impacts by N_2O was made taking an ozone depletion potential (ODP) of 0.017 as compared to CFC-11 ($CFCl_3$). The concept of Equivalent Effective Stratospheric Chlorine (EESC, expressed in ppt), applied by Struijs *et al.* (2010) and Ravishankara *et al.* (2009) was used to estimate the cumulative EESC reduction by N_2O in a scenario where all anthropogenic emissions of ODSs were halted in 2011 (Daniels *et al.*, 2010). This calculation takes into account both the different ODP and the atmospheric fate of the ODSs. The cumulative EESC reduction by N_2O between 2011 and 2050 was calculated at 1220 ppt yr, which is 6% of the total EESC reduction (Daniels *et al.*, 2010). Taking a DALY loss of 806 DALY/ppt/year (Struijs *et al.*, 2010), then yields a health loss of 24.2 DALY per kton of N_2O. From this the unit damage cost was calculated at 3 euro per kg N_2O –N (taking an economic value of 40 000 euro per DALY, as approximating to the value of VOLY used above). Some major sources of uncertainties are the choice of the N_2O reduction scenario and the dose–response relation for cataracts.

22.4 Costs of mitigation

Implementation of mitigation options generally involves cost to society. To some extent, this cost is a loss in benefits achieved by current N management practices as described in the previous section (e.g. benefits for agriculture). Although the exact distinction is difficult to make (so one must be careful not to double-count), this section mainly deals with the direct cost of mitigation options as a result of additional resources that are required when implementing these options (see also Section 22.2.3).

22.4.1 Mitigation options for air quality

The main sources of NO_x and NH_3 emissions are fossil fuel combustion and agriculture, respectively. Information on mitigation options for NO_x and NH_3 in Europe can be obtained from, e.g., the GAINS model (http://gains.iiasa.ac.at), which includes data on technical measures to reduce emissions from key sources. The GAINS databases on emission and cost parameters have been compiled over several years during national and industrial consultations accompanying preparation of the Thematic Strategy (CEC, 2005), NEC review process, and participation in the work of several UN Expert Groups on abatement technologies (Cofala and Syri, 1998; Klimont and Brink, 2004).

Measures that are available for mitigating NH_3 and NO_x emissions are all targeting emissions at the source. For NH_3 from agriculture, these include low N feed, low emission housing for cattle, pigs, and poultry, air scrubbing, covered slurry storage, low ammonia application of slurry and solid manures, incineration of poultry manure, and urea substitution (UNECE, 2007). It is important to stress that some of these options address only one ‘step’ in the emission cascade and so may move abated N from one compartment to the other, e.g. from housing to storage or from storage to land application. The benefit of single ‘compartment’ options is limited except for efficient land application that is at the end of the chain. Principally the measures should be applied in combination with, e.g., low emission housing with closed storage and low emission application.

NO_x emissions from energy combustion can be reduced via combustion modification (in-furnace controls, e.g. low NO_x burners), treatment of the flue gases (by selective catalytic reduction (SCR) or selective non-catalytic reduction (SNCR)), and measures in the transport sector (e.g. changes in engine design, fuel quality, after-treatment of the exhaust gas by various types of catalytic converters, on-board diagnostic systems, etc.). All of these technical measures are characterized by high reduction levels, ranging from 50%–60% for combustion modification to well over 90% for catalytic converters. In Europe, most of the potential for NO_x reduction is expected to be exhausted in the next decades (contrary to NH_3) as the currently implemented legislation, especially in the EU, requires installation of efficient technologies on stationary sources (see CEC, 2007a) and transport is asked to implement measures with reduction efficiencies over 97% (CEC, 2007b). Any remaining potential is very expensive, with marginal costs ranging from about 5 euro per kg NO_x-N in some Eastern European countries to more than 15 euro per kg NO_x-N in the Netherlands.

Figure 22.5 shows the emission reduction potential and marginal cost for NO_x and NH_3 for the EU27 in 2020 in addition to the measures already implemented under current legislation (based on the data in the GAINS model). The potential and marginal cost show great variation between countries.

It is important to note that some of the NO_x measures have potential for increasing emissions of NH_3 and N_2O (e.g. catalytic converters, fluidized bed combustion) or change the ratio of NO/NO_2 emitted which has implications for urban air quality. Although in recent years improvements have been made, owing to the sensitivity to some of these emissions, the issue should be monitored further.

Beyond the technical measures listed above, there are a number of so-called non-technical (management) measures having potential to reduce N losses to the air (or water), e.g. timing of application and increased grazing, energy conservation, traffic restrictions and speed limits. For agriculture, Oenema *et al.* (2007) and Velthof *et al.* (2009) characterize a much more exhaustive list of abatement measures that were implemented in the MITERRA-EUROPE model (see also Oenema *et al.*, 2011, Chapter 4, this volume). The costs of such non-technical mitigation options are generally more difficult to determine than the costs of technical measures.

22.4.2 Mitigation options for water quality

Nutrient emissions to surface waters can be reduced in different ways and at different stages of the nutrients’ pathway through the environment. A number of measures reducing emissions at the source, such as reductions in fertilizer use, reductions in livestock and reducing the N content of fodder, simultaneously improve water and air quality. Measures reducing the

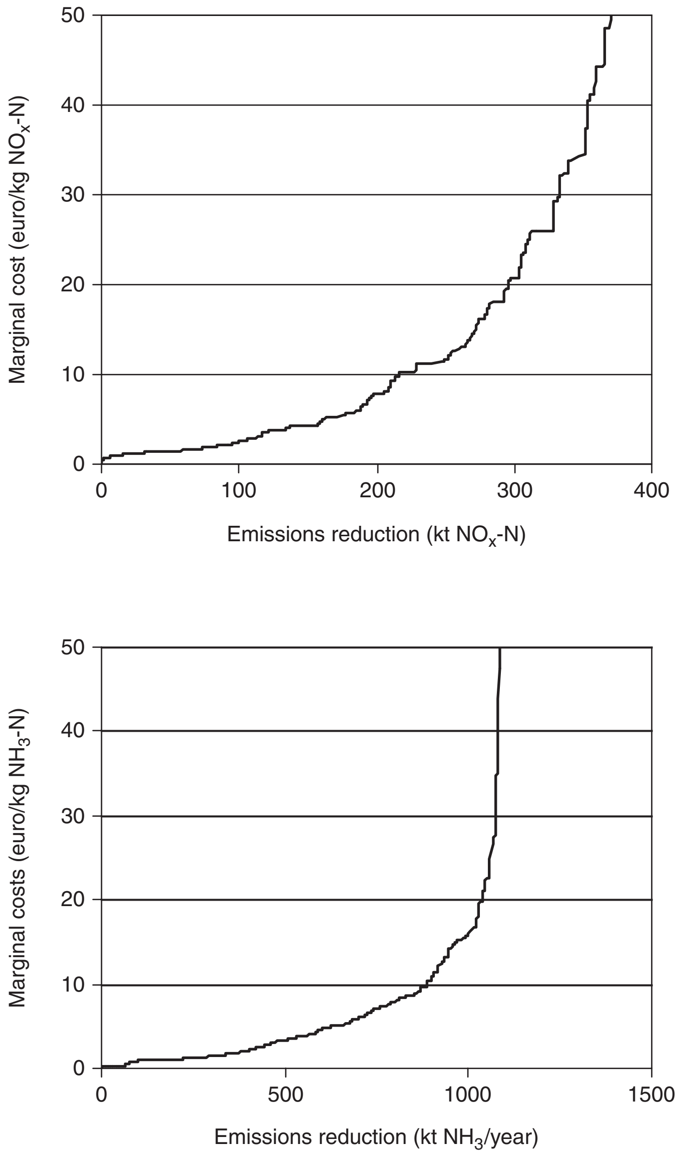

Figure 22.5 Cost curves for NO_x and NH_3 mitigation in EU27 countries in 2020; source: GAINS, current legislation scenario (2010 estimates).

emissions of N_r to the air (NO_x or NH_3) indirectly contribute to water quality because they result in a reduction in the deposition of N_r (see previous section). In addition, water quality improvements can be achieved by reducing N_r emissions from sewage treatment plants. In the EU these emissions are subject to the Urban Waste Water Treatment Directive, which requires substantial reductions in nutrients concentrations before the effluent is discharged into surface water (EC, 1991).

It is also possible to reduce the leaching of N_r from soil to water, e.g. by cultivation of catch crops and reducing discharges of N_r to surface waters by measures such as wetlands at river mouths along the coastal waters, manure-free zones along agricultural land and the use of helophyte filters (Gren *et al.*, 2008). Finally, once the N_r has been released into the surface waters, for some water systems it is possible to improve the water quality by removing the N_r from the system, e.g. by ecological fish-stock management (so-called bio-manipulation, see e.g. Lammens, 1999; Meijer, 2000; Lammens *et al.*, 2004) or by dredging.

The cost of measures to reduce the emission of N_r to surface waters largely depends on specific local circumstances; e.g. the price of land. Gren (2008) describes the calculated marginal cost of reducing N_r inputs to the Baltic Sea for a range of measures (Table 22.7).

Van den Broek *et al.* (2007) report that wet riparian buffer strips are more effective in reducing pollution than dry riparian buffers: if applied to larger regions wet buffers can decrease N_r emission to surface water by 15 kg N_r per ha at a cost of 37 euro per kg N_r making them a rather costly measure. Dry riparian buffers are even more expensive at an estimated 40 euro per kg N_r. Van der Bolt *et al.* (2008) report average additional cost of reducing N_r emissions to surface waters in the Netherlands of 70 euro per kg N_r for a package of measures including manure-free zones along surface water and of 45 euro per kg N_r for a package of measures including helophyte filters.

Figure 22.6 is an example of a cost curve for reducing N_r-leaching, based on an evaluation of the implementation of Action Plan for the Aquatic Environment II (Jacobsen *et al.*, 2004). Introduction of environmental sensitive schemes and reducing livestock intensity tend to be expensive measures, whereas wetlands and increased utilization of manure are relatively inexpensive.

Nitrates in drinking water can be reduced in concentration to prevent exceedance of limit values in drinking water at relatively low cost. Typical measures are mixing polluted water with clean water and biochemical water treatment. Alternatively, the infrastructure for drinking water collection can be adjusted (e.g. deeper extraction wells). Cost data are scarce but are expected to decrease with increasing scale of the drinking water production and treatment. Illustrative annual cost values are 0.5 euro per capita per yr for water treatment and mixing for the UK and the Netherlands where large aquifers are available (Pretty *et al.*, 2003; Van Beek *et al.*, 2006), 3 euro per capita per yr for Austria and Germany when extraction wells or drinking water infrastructure need adjustment (Ademsam *et al.*, 2002; Brandt, 2002) and 15 euro per capita per yr when new private wells are installed.

22.4.3 Mitigation options for N_2O (greenhouse gas balance)

There are two strategies to decrease N_2O emissions from agriculture (Oenema *et al.*, 2001):

- balanced N fertilization, i.e. increasing the N use efficiency together with a lowering of the total N_r input, and
- decreasing the release of N_2O per unit N_r from the nitrification and denitrification processes.

Increase of N-use efficiencies

Improving the N use efficiency reduces both direct N_2O emission from soils and indirect N_2O releases associated with ammonia emission and nitrate leaching. Measures to increase the N use efficiency in crop production systems, include adjustment of N_r application rate, method, and timing relative to

Table 22.7 Marginal (calculated) mitigation cost per kg N_r reduction of inputs to the Baltic Sea for a selection of emission reduction measures at sources and end of pipe (Gren *et al.*, 2008)

	Sewage treatment	Private sewers	Catch crop	Wetlands
	euro/kg N_r reduction to coastal waters			
Denmark	15–35	54–60	31–32	7–18
Finland	15–45	54–77	16–34	1–15
Germany	15–48	54–82	12–35	2–3
Poland	12–48	46–81	9–11	1–1
Sweden	15–79	54–81	5–40	8–290
Estonia	12–35	46–59	6–9	5–7
Lithuania	12–41	46–83	8	2
Latvia	12–49	46–70	15–22	7–10
Russia	12–67	46–115	17–21	10–15

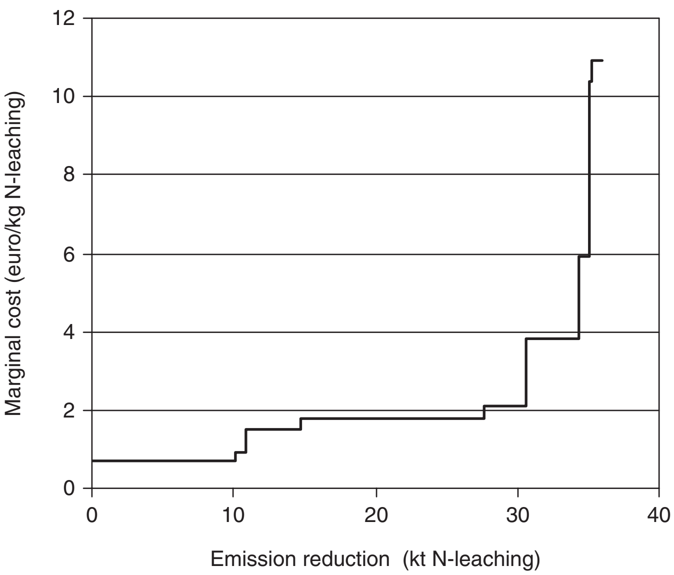

Figure 22.6 Cost curve based on the implementation of 'Action Plan for the Aquatic Environment II' in Denmark (Jacobsen, 2004).

the crop demand, use of soil and plant testing as a 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 are improved management of soils and crops. In general, there are no net costs or costs are low for these options, 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 at 5 US$ per ha cropland and of improved agronomy (i.e. agronomic practices to increase yields, such as changes in crop rotations) at 20 US$ per ha cropland (see also Jensen *et al.*, 2011, Chapter 3, this volume).

Decreasing the release of N_2O per unit N_r

Measures to reduce N_2O emission have to focus on avoiding application of N_r during conditions favourable for denitrification or to change them to create an environment less favourable for N_2O production (decrease the N_2O/N_2 ratio; also see Oenema *et al.*, 2001; Butterbach-Bahl *et al.*, 2011, Chapter 6 this volume). Options available include the following.

- Using ammonium based fertilizer (including urea) instead of nitrate fertilizer during wet conditions may significantly reduce N_2O emission (Clayton *et al.*, 1997; Velthof *et al.*, 1997; Dobbie and Smith, 2003; Jones *et al.*, 2005, 2007). This option is cost neutral but emissions of ammonia increase if urea is used without low-emission methods.
- 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 therefore decreases N_2O emissions. The costs of these types of measures are low, because they are based on correct timing of N application and choice of fertilizer type.
- Nitrification inhibitors delay the conversion of ammonium to nitrate (and possible denitrification of the produced nitrate). Fertilizer containing nitrification inhibitor costs about 1.5 to 2 times more than a common ammonium based fertilizer.
- Enhancing aeration of soils by proper drainage, irrigation and soil tillage and avoiding application of N during wet conditions reduce N_2O emission from soils. Associated costs are low.
- Removal of crop residues from the field. The costs are relatively high, because this 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. Costs are related to soil tillage and seed, and are higher than costs related to improved N management (see also Section 22.4.2).
- Proper mixing of the manure may decrease N_2O emissions from solid manure systems (Sommer, 2001). The costs of these measures are relatively low.

For further details on mitigation options for N_2O, see Supplementary materials.

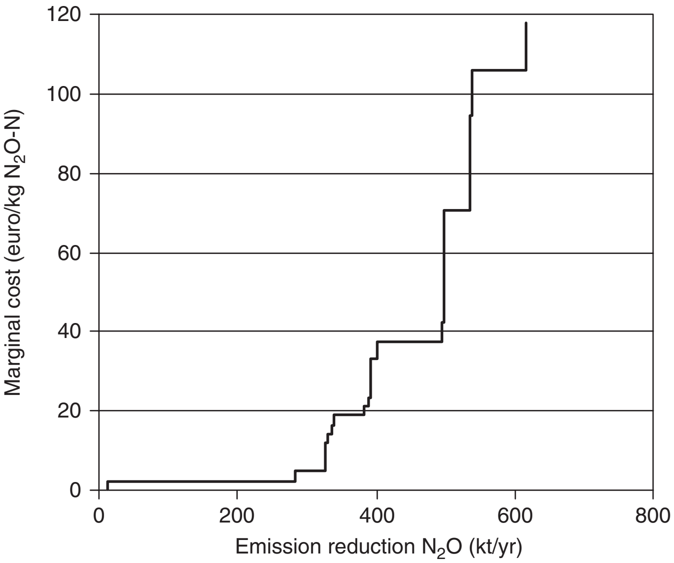

Figure 22.7 Cost curve for emission reduction of N_2O in EU27 for base year 2005; total emission is 1450 kt. Source: GAINS, current legislation scenario; Winiwarter *et al.* (2009).

The marginal abatement costs for N_2O calculated by GAINS (Figure 22.7) range from less than 1 euro per kg N_2O-N for balanced fertilization and adjusted tillage, up to 50 euro per kg N_2O-N for adjusted timing of fertilization and use of nitrification inhibitors (note that values can be up to 100 times higher than values per kg N-fertilizer in view of an emission factor for N_2O of about 1%).

22.4.4 Mitigation options for soil quality

Input of N affects soil organic matter content and quality, soil acidification and soil biodiversity and through this soil functions (see also Velthof *et al.*, 2011, Chapter 21 this volume): (i) storage, filtering, buffering and transformations of N, (ii) food and other biomass production, (iii) carbon sink and (iv) biological habitat and gene pool (Dise *et al.*, 2011, Chapter 20 this volume). In general, the adverse impacts of N inputs to soil quality of agricultural soils can be mitigated by modest adjustments of management of soil and crop residues.

In general, N_r has a positive effect on content and quality of soil organic matter agricultural soils (Glendining and Powlson, 1995). The results of Khan *et al.* (2007) and Shevtsova *et al.* (2003) suggest that in some circumstances N fertilization may enhance the mineralization of soil organic C. However, the apparent negative effect of mineral N fertilizer on soil organic C content may not only be related to enhanced mineralization, but also to differences in the input of crop residues. Options to maintain or increase the organic matter content in agricultural soils include the use of manures, growing winter crops, improved crop residue management, and reduced tillage (Smith *et al.*, 2008). The costs of such measures are relatively low (marginal cost of 0.05–0.1 euro per kg N_r for NW European countries, assuming an average N-rate of 100 kg per ha) and they may result in higher crop yields and quality.

Options to improve soil quality of natural soils are based on decreasing the N_r content of the soil, such as thinning and sod cutting, removal of the organic top layer, and choppering (see Diemont and Oude Voshaar, 1994; Niemeyer *et al.*, 2007). The costs of these measures per ha can be high, but in general are only applied on a local scale.

Liming is widely used to reduce acidification of agricultural soils. The average input of lime in NW Europe is 0.7 kg lime per kg N_r input. If it is assumed that the use of lime is needed to compensate the acidification caused by N-fertilizer; a rough estimate of the cost for lime use is then about 0.1 euro kg N_r. Also in natural systems, liming of soil may reverse the acidification processes (Beier and Rasmussen, 1994). However, Wolf *et al.* (2006) consider permanent liming of forests as an undesirable management option, because it increases the decomposition of soil organic matter leading to thin humus layers and a decrease in soil biota species. However, it can be beneficial as a once-only event in nutrient rich deciduous forest.

Options to restore loss of soil biodiversity are related to N_r input and include the use of manures, growth of winter crops and proper soil tillage, and restricted use of pesticides (Brussaard *et al.*, 2007; Kibblewhite *et al.*, 2008). The costs for measures are low and there may even be benefits for the farmers, as they may increase crop yield and quality. Smith *et al.* (2008) value the costs related to soil tillage and residue management at 5 US$ per ha per year, those related to nutrient management at 5 US$ per ha per year, and those related to agronomy (such as changes in crop rotation) at 20 US$ per ha per year.

22.5 CBA use in policy design and evaluation: case studies

22.5.1 CBA for support of European Air Quality Policy

Cost–benefit analysis of European air quality policy has built on the methodological framework developed under the European Commission-funded ExternE Project (Bickel and Friedrich, 2005). The first policy applications of this approach date back to the mid-1990s when it was applied to the EU's Acidification Strategy. Since then it has been applied in development of the EU's air quality directives, National Emission Ceilings Directive, the Gothenburg Protocol under the Convention on Long-range Transboundary Air Pollution (CLRTAP) and various other legislation concerning industrial emissions. Methods have been refined over time to factor in new research and the views of expert bodies including World Health Organization and working groups convened under CLRTAP. The analysis principally covers effects on human health, crops and building materials. Valuation of ecosystems has yet to be achieved, so ecosystem effects are described only in terms of critical load exceedance.

Figure 22.8 illustrates results for the EU's Thematic Strategy on Air Pollution (Pye, *et al.* 2008), which feeds into the revision of the National Emission Ceilings Directive. The figure shows for each country the ratio of benefits to costs using a conservative estimate of health benefits. The scenario for which these data

Table 22.8 Estimated costs and benefits of applying best available emission control techniques at the 10 plants in the EU26 with the largest combined SO_2 and NO_x baseline emission (Barrett and Holland, 2008)

Rank	Country	Plant	Electrical capacity, MW	NO_x emission kt/yr	NO_x benefit, €M/year	Total benefit, €M/year	Total cost €M/year	Benefit–cost ratio
1	Bulgaria	Maritsa II	1450	58	103	985	101	9.79
2	Spain	Puentes	1400	19	47	1357	122	11.11
3	Greece	Megalopolis A	1400		3	285	70	4.08
4	Spain	Teruel	1050	31	77	497	65	7.62
5	Poland	Belchatow	4340	39	147	885	290	3.05
6	Bulgaria	Maritsa I	200	10	26	282	26	11.03
7	Poland	Patnow	1200	11	39	521	100	5.22
8	Spain	Compostilla	1312	31	85	340	107	3.19
9	UK	Cottam	2008	26	74	505	137	3.69
10	UK	Drax	3960	54	198	338	191	1.77

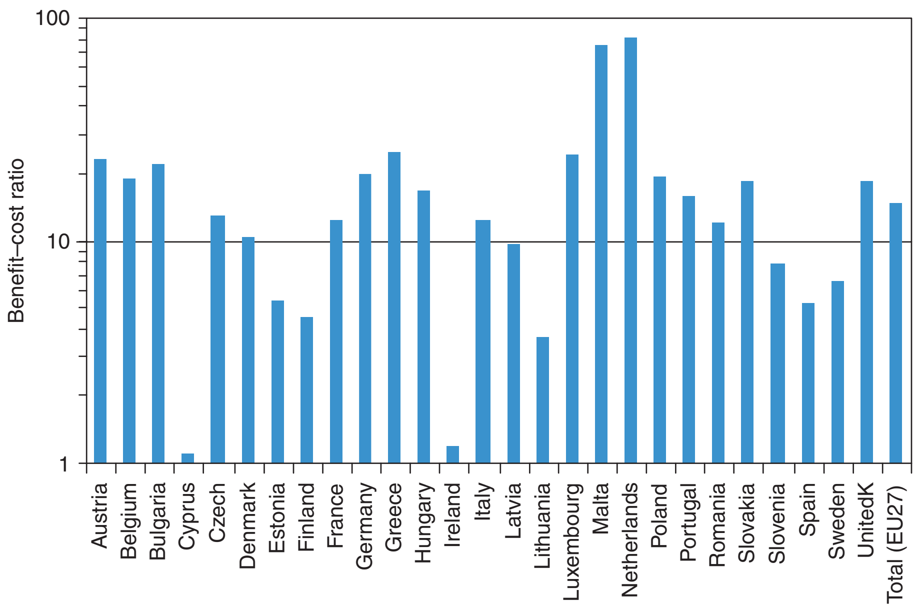

Figure 22.8 Illustration of effect of EU Thematic Strategy on Air Pollution (EU) on the ratio of health benefits and cost of measures. Results show benefit–cost ratios for each country under a scenario taking the CAFE-low case for benefit estimation. In all countries the ratio is greater than 1, which implies that a net benefit will be achieved.

were obtained seeks to meet the environmental quality objectives of the Thematic Strategy in the most cost-efficient way, according to the GAINS model (IIASA, 2008). The results presented take account of the EU's Climate and Energy Package, supplementing the initial analysis for the Commission for which the benefit–cost ratio was lower (demonstrating the clear co-benefits of combined climate and air quality policies). It is clearly demonstrated that the Thematic Strategy policy scenario is estimated to generate significant net benefits for the EU relative to costs. Significant variation in benefits per unit cost is clear across the EU (reasons for which are discussed in more detail below). The application of Monte Carlo analysis and sensitivity analysis demonstrated that the principal conclusion drawn from the quantification, that benefits of the policy would exceed the costs, was robust.

Damage costs from different studies are often of limited comparability because of variations in views on what damage should be quantified, dose–response and valuation functions, release of and exposure to air pollutants in different countries and scale. For a consistent set of assumptions, the marginal damage estimates per tonne NO_x release can vary by a factor 20 (530–9600 euro per tonne), depending on the country or sea region in which the emissions occur (Methodex, 2010). Given the dominance of health impacts in this analysis the differences between countries largely reflect the probability of someone being exposed to emissions from a particular source. Emissions from countries in central Europe that are surrounded by fairly populated areas for hundreds of kilometres all round are therefore linked with greater damage than countries around the geographical fringe of Europe. Any marginal damage estimates provided without explicit mention of such assumptions are clearly difficult to understand. Similar results can be obtained for ammonia where damage costs in EU25 Member States range between 700 and 30 000 euro per tonne.

It is therefore concluded that despite the presence of significant uncertainties CBA is a useful tool to support European air pollution policy development. A further practical example of the application of CBA is using cost–benefits ratios to select power plants where application of emission control techniques are expected to generate highest social benefits for the EU (Table 22.8; taken from Barret and Holland, 2008).

22.5.2 Integrated nutrient management of the Baltic Sea

Policy background

Damages from eutrophication in the Baltic Sea due to excess N_r and phosphorus loads have been documented since early 1960s by a number of different studies (see, for example, Wulff *et al.*, 2001). The affected countries also showed concern by, among other things, the establishment of the administrative body Helcom in charge of policies for improving the Baltic Sea since 1974, and ministerial agreements in 1988 and 2007 (Helcom, 1993; Helcom, 2007). However, approximately 20 years after the meeting in 1988, the agreed level of nutrient reductions in 1988 is far from being reached. One important reason for the hesitation to reduce nutrient loads to the Baltic Sea is likely to be the associated costs, which now start to increase at a higher rate than earlier since the low cost options, such as improvement in nutrient cleaning at sewage treatment plants located at the coastal waters of the Baltic Sea, have been implemented in several countries. Another reason is the differences among countries in perceived benefits from nutrient reduction. Furthermore, a successful implementation of an international agreement requires a perception of fairness by involved stakeholders (Carraro, 2000; Bérubé and Cusson, 2002; Lange *et al.*, 2007).

In order to calculate net benefits and compare these with different fairness criteria integration is needed of (i) N and phosphorus transports, (ii) upstream and downstream located abatement measures, and (iii) economic and fairness conditions. Although there is a large literature on net benefits from international environmental agreements, there are few studies considering this together with fairness outcomes and with several interlinked pollutants and abatement options. Existing evaluations of international agreement have been made mainly for energy policies (Carraro and Buchner, 2002; Lange *et al.*, 2007; Dannenberg, 2008).

Modelling and data retrieval

The typical approach for evaluations for energy policies has been to calculate net benefits of mitigation and adaptation strategies and to compare these with different fairness criteria. Using this approach and cost minimization for the Baltic Seas takes into account a number of different abatement measures which either reduce nutrient loads from sources or act as sinks for nutrients. Examples of the former are the use of selective catalytic reductions at combustion sources, livestock reductions, and decreases in use of N fertilizers. 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. 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. Unless otherwise stated, all data and calculations are found in Gren *et al.* (2008). Given all assumptions, the calculated total nutrient loads of approximately 830 kt of N_r and 40 kt of phosphorus, which come relatively close to the estimates obtained in Helcom (2004) (for further details see Supplementary materials).

Net benefits and fairness

Although there is a general consensus on the requirement of fairness for truthful implementation of cleaning plan, there is less agreement on the operational definition of fairness. Usually, a distinction is made between the processes of reaching agreements and the outcome of the agreements (Carraro, 2000; Grasso, 2007). When focusing on fairness, two principles can be distinguished; egalitarian and equity. The egalitarian principle rests on equal human rights, where citizens have the right to, for example, the same amount of emission of N and phosphorus. The equity principle, based on the capability approach suggested by Sen (1999), relates financial burdens of actions to the agents' ability to meet them. Based on these two principles of fairness with respect to allocation of cleaning among countries, two criteria are included: emission per capita and related to gross domestic product (GDP).

Calculated net benefits from a cost effective achievement of Helcom Baltic Sea Action Plan (BSAP) and fairness outcomes are presented in Table 22.9.

The results presented in Table 22.9 indicate a total net gain from the implementation of the Helcom BSAP, but also that the net benefits are unevenly distributed among the countries. However, positive net benefits for all countries can, in principle, be obtained by a suitable choice of international policy instruments. Under a nutrient trading scheme, choice of distribution of initial permits, which implies capital transfers, can be adjusted in order to affect countries' net benefits. This will also have impact on the outcomes of the fairness criteria, which show significant differences in load per capita and load per GDP. For example, Poland has relatively low nutrient loads but also faces the largest net loss from the abatement programme. This case study points to the need of integrated assessment of net benefits and fairness criteria for truthful implementation of international water management agreements.

22.5.3 Cost of implementation of the Nitrate Directive in Denmark

Measures and costs to fulfil the Nitrate Directive in Denmark

The implementation of the Nitrate Directive in Denmark through the Action Plan for the Aquatic Environment II

Table 22.9 Net benefits and fairness under cost effective achievement of the Helcom BSAP

	Net benefits		Fairness criteria			
			Load/capita		Load/1000 I$ GDP	
	Mill I$	I$/capita	Kg N	kg P	kg N	kg P
Denmark	816	177.4	9	0.12	0.26	0.003
Estonia	−111	−82.8	31	0.51	1.67	0.027
Finland	507	96.4	9	0.20	0.26	0.006
Germany	513	155.5	12	0.08	0.38	0.003
Latvia	−163	−71.2	13	0.63	0.84	0.041
Poland	−1752	-45.9	6	0.23	0.40	0.016
Sweden	1354	149.2	8	0.09	0.24	0.003
Russia	−86	−9.7	8	0.26	0.64	0.020
Lithuania	−270	−79.2	17	0.33	1.12	0.021
Total	*808*	*10.6*	*8*	*0.22*	*0.41*	*0.011*

In order to account for differences in purchasing power among countries, cost and benefit estimates are adjusted by the purchasing power parity (PPP) index and measured in international dollars, I$. The PPP index reflects the purchasing power of a dollar in each country, and varies between 0.7 and 1.9. This adjustment implies an upward adjustment of net benefits in countries with PPP>1 and a downward adjustment when PPP<1.

(Action Plan II) has been accepted by the EU. Based on the results from the technical evaluation of Action Plan II, the cost effectiveness of each measure is calculated (Jacobsen, 2004). The total cost connected to the area related measures (top four measures in Table 22.10) was 27 million euro per year. The reduction in cost compared to expectations is mainly due to a smaller land area with voluntary agreements. The area related measures carry half the costs, but only 16% of the reduction in N-leaching. It should be noted that the area related measures serve a range of purposes which have not been valued, such as lower phosphorus loss, lower pesticide usages and biodiversity.

One of the most important farm related measures has been lowering the legal N_r application standards by 10% which is discussed below.

The cheapest measures are (1) construction of wetlands, (2) better utilization of N_r in animal manure and (3) changes in feeding. The Environmental Sensitive Schemes (ESAs) and lower animal density on farms are among the most expensive measures when the cost is related to only the reduction in N-leaching. The area-related measures have not achieved objectives, mainly due to the low area involved in their application. On the other hand the reduction in N-leaching due to the farm related measures has achieved the expected level of control and on top of this come the additional measures at the farm level, which ensure that the total aim has been achieved. The total cost is 70 million euro and the cost effectiveness is approximately 2.0 euro per kg of reduced N leaching.

Lower N application – costs and considerations

When trying to estimate the costs of reducing N_r applications there is a need to look both at the change in yield and quality as well as the implications for the value to the farmer. The N application standard was introduced in Denmark in the late 1980s, with maximum application equal to the economic optimum in 1991. This in itself reduced the N-application as some farmers applied more than required for the optimum. In 1998 the application standard was reduced to 10% below the economic optimum of 1991. In the year 1997/98, 38% of all farms applied 20 kg N per ha per year below the standard. These farms were organic farms, but also dairy farms where standards were not binding. This percentage dropped to 10% in the year 2000. The associated area is 1.9 million ha as compared to 2.3 million ha where the application standards were in place in 1997/1998.

Another element of the Danish N-policy is that total national N-application is capped to ensure that the national application rate will not rise if, for example, crop prices increase and/or fertilizer costs fall.

The total cost of a reduction of 10% in N-standards was estimated using a sector-model at 23 million euro; 10–15 million euro due to lower yield and 7–10 million euro due to loss of crop quality. These model estimates allow change of crop rotation when this is profitable (for more detail see Jacobsen *et al.*, 2004). Field trials for cereals have shown that the protein content drops by 0.2% per 10 kg N reduction of the N_r-application. The economic cost of lower protein varies from crop to crop; for bread wheat the cost is fairly high, for barley and export wheat cost is average, and low for forage crops.

For development of Danish N-policies environmental gains were not monetized for comparison against economic loss for the farmers. The approach is aimed at finding the most cost-effective measures to reach the target. The reduction of N standards to 10% below the economic optimum has increased the incentives to optimize handling of all N_r resources. Together with the required utilization of animal manure there is a large incentive to optimize N_r usage at the farm level. Changes in feeding have reduced the N-leaching more than expected and while implementation costs are limited.

Table 22.10 Cost effectiveness (euro per kg N-leaching) for the different measures in Action Plan for the Aquatic Environment II

	Area	Reduction N-leaching	Total cost	Cost effectiveness
	1000 ha	kt N	million euro	euro per kg N
Wetlands	2.9	0.8	0.7[c]	0.9
ESA-areas	25.7	0.7	7.7	10.9
Afforestation	14.2	0.8	4.7[c]	5.9
Organic farming	111.5	3.7	14.0	3.8
Changed feeding		3.8	5.7	1.5
Lower livestock density [a]		0.14	1.5	10.4
Catch crops (6%)		3.0	6.4	2.1
Increased utilization N in manure (15%) [a]		10.1	6.7	0.7
Reduced N-standards (10%)[a]		12.9	22.8	1.8
Sum [b]		35.9	70.2	2.0

[a] In the technical evaluation the effect of these measures has not been divided into the effect of each measure, which is why the estimation here is somewhat uncertain.
[b] Changes in land use and animal production are not considered.
[c] Annualized using a 4% discount rate and infinite lifetime.
Source: Jacobsen (2004).

22.5.4 Cost of implementing Nitrate Directive for Dutch agriculture

Policy development since 1998

Between 1998 and 2006 the Netherlands used the Mineral Accounting System (MINAS) to implement the EU Nitrates Directive. MINAS was a system based on farm gate balances and levies, that did not strictly implement the EU application standard for N in manure of 170 kg per ha per year. For this reason the EU Court of Justice ruled in 2003 that MINAS had to be replaced by a system of more rigid application standards. Another reason was the persisting exceedance of the environmental target in groundwater of 50 mg/l nitrate in sandy and clayey soils in use for agriculture, in spite of apparent decreases of N_r surpluses and nitrate concentrations since 1998. Implementation of application standards for N included a time schedule for gradual tightening of standards based on environmental demands and economic feasibility. In the new system arable farms could accept less manure and part of the dairy farms had to dispose of manure, causing an increase of manure supply. As a result, costs to dispose of manure went up which particularly affected land-less intensive livestock farms. While arable and dairy farmers had several readily available farm measures to deal with their problems, the most feasible solution for pork and poultry farmers was manure processing and export. However, this solution was costly and stimulation of innovation required a subtle and time consuming interaction between farmers (cooperative), commercial manure processors, research and national and EU policies (criteria for export and acceptance of processed low carbon manure as mineral fertilizer when complying with legal application standards for total N_r).

Arable and (non-greenhouse) horticulture

For clay soils, application standards were set for the period 2006–2009. For crops on sandy soils where the fertilizer recommendation leads to exceedance of 50 mg/l nitrate in shallow groundwater, establishment of legal standards was postponed to 2007.

Modelling showed that costs for arable farming on clay soils mainly result from a reduction of income linked to limitations on manure use and not from lower crop yields due to N_r-shortage. For arable farming on sandy soils, with lower application standards and less opportunity to use manure, a reduction of application standards by 30% in 2009 relative to 2006, costs are similar to those on clay soils, but mainly caused by yield loss due to N_r-shortage. Typical compensating measures to reduce costs are the use of green manure and the application of livestock manure in spring instead of late summer. Precision application of fertilization is also an option, but rather costly and therefore more applicable for horticulture. Costs of measures were also expressed as a reduction of farm income (Table 22.11). This reduction is fairly low, but not irrelevant as profit margins for arable farming also tend to be low. The total annual national cost resulting from a 30% reduction of application standards for arable farming and horticulture on sandy soils (about 200 000 ha) was estimated at 4 million euro (20 euro per ha), which is a modest amount compared to the total cost of production or administrative costs of N-policies. Furthermore it was found that costs were restricted to a small group of farm types, in particular farms with no possibility to use manure (e.g. horticulture on sand). Both findings suggest that there is still scope for substantial and cost-efficient reduction of N_r use in arable farming.

Table 22.11 Examples of reduction of economic yield due to lowering of application standards in 2009 relative to 2006

Sector	Soil type	Reduction of application standards, 2009 relative to 2006	Example of reduction of application standard kg/ha	Costs without compensating farm measures euro/ha	Costs with compensating farm measures euro/ha (% income)
Arable	Clay	0%	Winter wheat: 245→220	15–35	5–20 (1%–4%)
	Sand and loess	−30%	Winter wheat: 19→160	0–55	0–40 (0%–5%)
Vegetables	Clay	0%	Brussel sprouts: 320→290	0–20	0–5
	Sand	−30%	Brussel sprouts: 290→275	355–490	65–165 (3%–6%)
Flower bulbs	Sand	−30%	Tulip: 200→190	250–1075	0–295 (0%–18%)

Source: PBL (2007).

Table 22.12 Effect of N_r intensity on economic yield of dairy farms (inferred from Van den Ham *et al.*, 2007)

Milk production	500 t/yr or 12.5 t/ha			700 t/yr or 15.0 t/ha		
Nitrogen intensity	**High N**	**Low N**	**Difference**	**High N**	**Low N**	**Difference**
Fertilizer N_r (kg/ha)	164	89	75	195	140	55
Manure N_r (kg/ha)	272	242	30	339	284	55
Feed N_r (kg/ha)	91	92	1	119	120	1
Soil surplus N_r (kg/ha)	226	147	79	256	168	88
Milk yield (euro/100 kg)	30.5	30.5	0.0	30.2	29.5	0.7

Dairy

Negative economic impacts of the new policy for the dairy sector are modest. Costs can amount to 5–45 euro per ha and occur mainly on intensive dairy farms on sandy soils that apply grazing. These farms may have to buy additional fodder, the cost of which is not compensated by savings on fertilizer. However, most dairy farms can take measures to reduce these costs, like increasing their acreage of silage maize. It was estimated that due to the introduction of the new system the dairy sector had to dispose an additional 2.2 million tonnes of manure in 2006 at a cost 15–20 million euro (1.5–2 euro per ha).

Monitoring results also indicate that there is no clear relationship between N_r intensity and economic yield from milk production (Table 22.12). For moderately intensive dairy farms there is no effect at all and for intensive farms economic loss is about 2%.

Intensive animal husbandry

The major source of costs from limiting N_r application for intensive animal husbandry is storage and transport of manure (Figure 22.9; Van Grinsven *et al.*, 2005). Introduction of the system of application standards, including a strict limit for N in manure (170 kg/ha for arable farms and dairy farms (10%) with no derogation and 250 kg/ha for dairy farms with a derogation (90%) increased manure costs for the pig and poultry sector by about 10 euro per tonne, amounting to national costs of 90 million euro in 2006; PBL, 2007). About half of these costs are not related to storage and transport but to additional remuneration to arable farmers for accepting manure (these transfer costs are not considered for national assessments).

22.6 Synthesis and discussion

22.6.1 Costs and benefits of N on human health, ecosystems and greenhouse gas balance

The results of the monetized environmental impacts for the different N_r-compounds are summarized in Table 22.13 as unit damage costs, i.e. the value of the impact, per unit of N_r, on human health, ecosystems and greenhouse gas balance. The values are presented as a range given the large uncertainties surrounding these damage cost estimates. For example, the relationship between damage and emission levels is in most cases non-linear therefore the unit damage costs (which in fact are the slopes of response function) will depend on the level of N_r emissions. Furthermore, unit damage costs vary between countries by a factor 20 to 100 due to differences in dispersion, exposure and mitigation between countries (see Sections 22.3.3 and 22.3.4). Ideally, the estimates are presented as a function of both biophysical and human population characteristics that significantly affect the size of the impact. Moreover, the estimates are based on different methods, adding to the complexity of direct comparison of

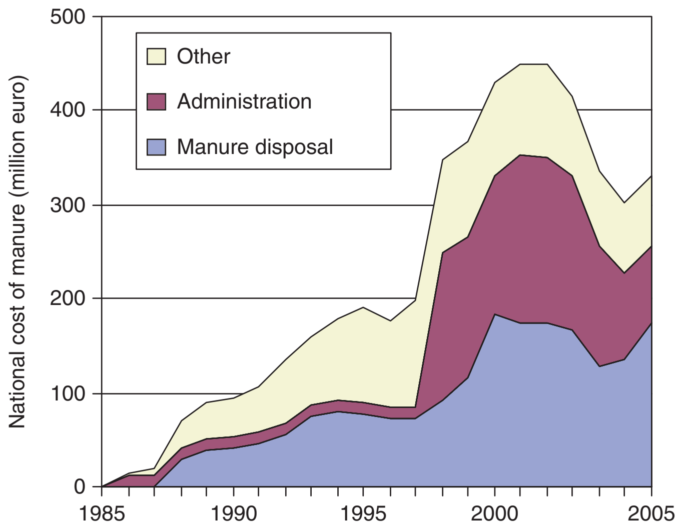

Figure 22.9 National costs from fertilizer and manure policies for agriculture in the Netherlands (manure disposal costs are net result of additional costs for animal production and additional income for arable production).

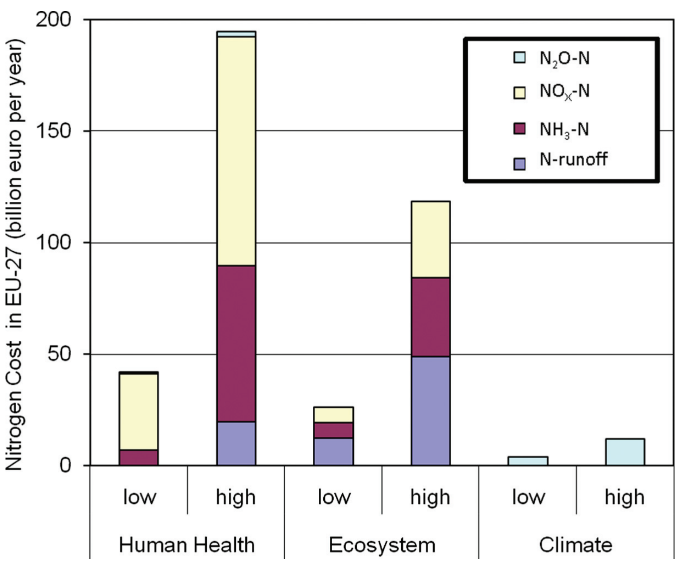

Figure 22.10 Low and high estimates of total social damage in EU27 as a result of environmental N-emissions in 2000.

the value estimates. It is important to appreciate that there are uncertainties on both sides of the cost–benefit equation. While valuation of the costs of measures to reduce the emission of different N_r compounds and mitigate their effects is fairly well developed, particularly for the air compartment, there is a tendency for costs to be overestimated as they do not account for refinement of existing approaches and the development of new ones by industry. The importance of this tendency varies with the type of policy under investigation. It can be particularly large for flexible mechanisms such as the use of emission ceilings. Quantification of the impacts of N_r and associated values is also affected by uncertainties, with particular problems affecting valuation of mortality and quantification of ecosystem impacts.

Using economic efficiency as an evaluation criterion, the marginal abatement cost for a specific N_r-compound should not exceed the associated marginal social benefits in terms of avoided damage costs presented in Table 22.13, unless it is considered that there are significant additional elements that remain unquantified, which can often be the case. Current policy scenario studies, however, commonly consider marginal abatement and mitigation costs exceeding the values presented in Table 22.13. Only the mitigation costs for N-leaching used by Jacobsen (2004) and Gren *et al.* (2008) are somewhat lower and do not exceed the social benefits of decreased environmental damage.

Aggregating the average unit damage costs presented in Table 22.13 across the EU27 using emission data provides an indication of the total damage due to the emission of N_r. This is presented in Figure 22.10. Again a lower and upper bound is presented in order to properly reflect the uncertainty underlying the damage cost estimates.

Accounting for the impacts of the emissions in 2000 of N_2O, NO_x, NH_3 to air and N to water, the total annual N-damage in the EU27 ranges between 70 and 320 billion Euro. This corresponds to a welfare loss of 150–750 euro per capita, which is in turn equivalent to 0.8–3.9% of the average disposable per capita income in the EU27 in 2000 (Eurostat 2010). About 60% of these damage costs are related to human health, 35% to ecosystem health and 5% to the effects on the greenhouse gas balance.

Despite these difficulties, some provisional conclusions can be drawn, namely the following.

(i) Health impacts of airborne pollution contribute most to social cost of N_r.
(ii) Social cost of environmental damage by airborne pollutants NO_x-N and NH_3-N are similar, but those for NH_3-N are more uncertain.
(iii) Social cost of damage to ecosystems caused by N-runoff appears to be broadly similar to that by airborne N_r (atmospheric deposition).
(iv) The social cost of the damage to aquatic ecosystems by N-emissions to water is higher than that to public health. This is as expected as nitrate pollution of drinking water is strictly regulated and most tap water is purified or blended.

22.6.2 Costs and benefits for agriculture

For illustration of the scope for improvement of N-management in agriculture it is more meaningful to express N-damage costs and benefits per unit of N applied to agricultural land. This is possible by combining unit damage costs for N-compounds in Table 22.13 with emission factors for N-compounds per unit of N application (see e.g. Velthof *et al.*, 2009). Some indicative results are shown for Calcium Ammonium Nitrate (CAN) application (CAN is the most used chemical fertilizer in Europe) to arable land (Table 22.14). Because of the low emission factors for airborne N-compounds from CAN, unit costs by N-emissions to water are the most prominent damage items.

Table 22.13 Emissions of N_r in EU27 and estimated ranges of unit damage costs for the major N_r pollutants and, between brackets, single values inferred from studies used in this assessment

	Emission-EU27[a]		Health	Ecosystem	Climate	Total
	Tg N_r	% agric	euro/kg N_r	euro/kg N_r	euro/kg Nr	euro/kg N_r
N_r to water	4.9	60	0–4 (1[b])	5–20 (12[d])		5–24 (13)
NH_3-N to air	3.5	80	2–20 (12[c])	2–10 (2[e])		4–30 (14)
NO_x -N to air	3.4	10	10–30 (18 [c])	2–10 (2[e])		12–40 (20)
N_2O-N to air	0.8	40	1–3 (2[f])		5–15 (9[g])	6–18 (11)

[a] EU27 Emissions for year 2000 based on various sources (e.g. EMEP, MITERRA)
[b] Health damage from nitrate in groundwater based drinking water based on Grinsven *et al.* (2010). Lower limit for unit damage costs for health impacts of NO_3 (colon cancer)
[c] Based on unit damage costs damage for airborne NO_x (20 euro/ kg N_r) and NH_3 (12 euro/kg N_r) from ExternE (2005) after conversion of results per mass of pollutant to mass of N_r in pollutant. Range arbitrarily set at ± 10 euro/ kg N_r for both NO_x and NH_3. With respect to NH_3 the lower bound reflects the present debate over the importance of health impacts from ammonium in airborne particulate matter.
[d] Upper bound based on WTP for a 'healthy Baltic' from study of Söderqvist and Hasselström (2008) and assumption in Gren *et al.* (2008) that damage can be repaired by 50% reduction of N-load to Baltic Sea. Lower bound arbitrarily set at 25% of upper bound.
[e] Ecosystem damage by deposition of NH_3 and NO_x on terrestrial ecosystem. Lower bound based on the EU NEEDS project (Ott *et al.*, 2006) representing the cost for restoring biodiversity loss due to N_r. Upper bound arbitrarily set at 5 times lower bound as a possible value when using an ecosystem service approach (uncertain share of N_r).
[f] Increased incidence of skin cancers and cataracts from depletion of stratospheric ozone. Unit damage cost is inferred from a global LCA study by Struijs *et al.* (2010).
[g] Climate damage based on contribution of N_2O-N to greenhouse gas balance and CO_2-price. Uncertainty range based on variation of CO_2-price since 2005 between 10 and 30 euro/t.

Table 22.14 Long term social cost of adverse N-effects per kg of CAN-fertilizer application

N-effect	Min	Max
Nitrate groundwater	0.0	1.4
N-load surface water	0.3	4.0
NH_3-emission to air	0.0	0.9
NO_x-emission to air	0.0	0.2
N_2O-emission to air	0.1	0.3
Total damage	0.4	6.8

Unit damage costs for N_2O are small compared to other cost items, implying that N-policies for agriculture should not focus on reduction of emissions of N_2O. In addition there are damage costs related to emissions of NO_x, NH_3 and CO_2 during manufacturing of chemical fertilizer; in the range of 0.1–0.3 euro/kg N_r (Von Blottnitz *et al.*, 2006).

Unit (short-term) benefits of N-fertilizer for producing bulk commodities such as cereals, potatoes, sugar beet, and milk range between 0.3 and 3.3 euro per kg CAN-N (Section 22.3.2). This range is rather narrow compared to the maximum range of damage costs of 0.4–6.8 euro per kg CAN-N (Table 22.14). The ratio of marginal private benefits over marginal environmental cost of an additional kg of N then would then be around 0.5 (taking ratios of lower bounds and upper bounds respectively, assuming that WTP for agricultural products and environment in the EU are correlated). Taking into account long-term benefits of N to secure soil N-availability could nearly double the N-benefits, and would raise the ratio to a value around 1. Upscaling of marginal damage cost to EU27 gives an annual social cost of between 20 and 150 billion euro, as compared to an annual benefit of N-fertilizer for farmers between 10 and 50 billion euro, or 20 to 80 billion euro when including long-term N-benefits (Section 22.3.2).

Results suggest that for the present levels of N-fertilisation, the marginal environmental costs of the use of CAN tend to be close to the marginal agricultural benefit. As N-emissions and social impacts increase proportionally with the use of CAN, while effects on crop yield level off (see Section 22.3.1), the risk of externalities exceeding crop benefits will tend to increase with higher inputs. However, it should be stressed that the upper bounds of the environmental costs are indicative and have a lower probability of occurrence than the empirically based upper bounds of the agronomic benefits. Von Blottnitz *et al.* (2006) estimated environmental damage (externalities) from fertilizer application at about 0.5 euro/kg N, which corresponds to the lower bound of the range presented here (Figure 22.11). The value by von Blottnitz *et al.* (2006) is low mainly because they used low cost estimates for N-runoff based on Pretty *et al.* (2003; see Section 22.3.4).

When part of the N-addition is in the form of manure, the difference between externalities and net crop benefits will increase in view of the higher emission factors for ammonia (up to 70%), and the lower fertilizing efficiency of N in manure as compared to chemical fertilizer. In view of the high unit damage cost for ammonia the use of manure-N without applying far reaching low emission techniques, therefore, would often be not beneficial for society. To a lesser extent this is also true for use of urea fertilizer, that loses around 15% of N as ammonia.

First estimates of how much N application rates should be reduced are obtained by comparing the social optimal N-rate (SONR) to the farm (private) optimal N-rate (PONR; see equations given in Section 22.3.1). For winter wheat in Germany (data from Henke *et al.*, 2007), and oilseed rape (using data by Sieling and Kage, 2008) SONR was between 35 and 90 kg/ha lower than the PONR. This difference corresponds rather well to results by Brentrup *et al.* (2004) who found a difference of 50–100 kg/ha by applying LCA using winter wheat data from the Broadbalk Experiment at Rothamsted in the UK.

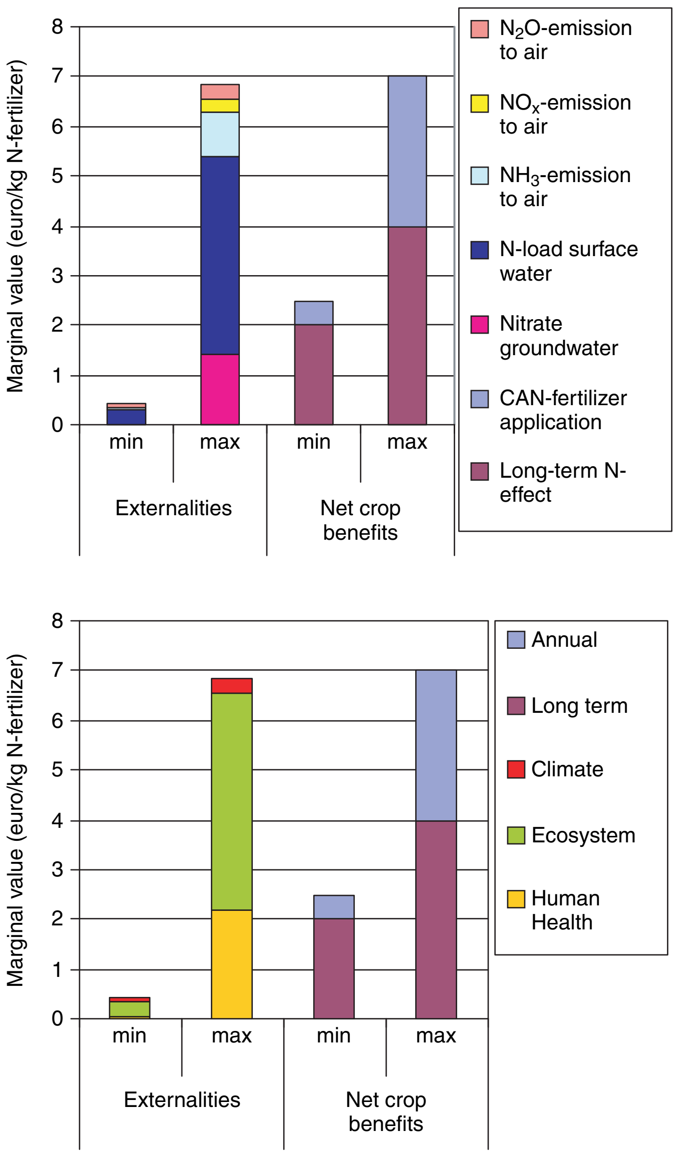

Figure 22.11 Comparison and breakdown of low and high estimates of marginal costs and benefits per kg of applied N at present levels of N-fertilizer (as Calcium Ammonium Nitrate) application for arable agriculture on sandy loam.

Although these results are only indicative, they illustrate the extent of the social costs caused by present N_r-fertilization rates in agriculture and the benefits achievable when reducing current N_r-application levels accounting for the environmental costs. However, one also has to keep in mind the social importance and value attached to existing farming systems in Europe. N_r-benefits are incomplete as social concern about food security and the socio-economic position of farmers and rural communities are not considered here, as well as the benefits of affordable agricultural products for industry, retailers and consumers. For example, applying SONR for winter wheat in NW Europe would cause a loss of grain of 1–2 tonne/ha. This production loss of 20% would need to be compensated, e.g. by developing more N-efficient wheat varieties or increasing wheat production in SE Europe or other parts of the world.

22.6.3 Discussion and future challenges

Air pollution

CBA, using indicators such as 'unit damage costs per tonne emission' and 'benefit–cost ratio', are increasingly used to evaluate and adjust air pollution policies (see Sections 22.5.1 and 22.5.2). Presently, the EU CAFE programme is the most integrated operational approach for using CBA to support integrated N policies, but this approach does not include emissions and effects of N_r in soil and water, and focuses primarily on human health impacts (partly because of the high level of concern over them). Although cost–efficiency and cost–benefit results from the CAFE procedures are the foundation for setting National Emission Ceilings for SO_2 and NO_x, they are not for ammonia. In view of the linkage between the ceiling for ammonia and cost of agricultural production, setting this ceiling is primarily based on political negotiation. The resulting negotiated ceilings still cause massive exceedance of critical N deposition levels for ecosystems.

ExternE used a 'revealed preference' approach to determine the value of acidification and eutrophication effects. This approach is based on the assumption that the decisions policy makers have made in the development of the Gothenburg Protocol under the UNECE Convention on Long-Range Transboundary Air Pollution and of the EU's National Emission Ceilings Directive provide a proxy for the the social value of acidification and eutrophication. Whilst this method can help to assess consistency between policies, the use of such numbers assumes that policy makers are fully informed about all impacts and their implications for social welfare. These not only include environmental damages, but also a number of other factors (e.g. concerns over competitiveness and employment) come into play when decisions on permissible levels of air pollution and associated damage are under consideration. All these social costs and benefits will affect the apparent relationship between damage and the costs of avoiding it.

In considering how policy makers should react to the current problems with eutrophication it is worth referring back to the problem of acidification in the 1980s. This was seen as being linked only to acidification and its effects on ecosystems (particularly forests and freshwaters) and building materials (particularly for monuments). Now, however, the problem has expanded and impacts on human health are considered the dominant driving force for regional European air pollution policy. Concern about ecosystem damage is increasingly linked to N_r deposition and eutrophication, rather than to acidification. It could be that the policy response to health impacts is little different to an optimal response for ecosystem impacts, in which case the omission from valuation of the latter may be of little consequence. In terms of environmental protection it would be useful to conduct a qualitative assessment of the type of impacts that the health based policies could and could not mitigate, and the extent to which the areas that benefit most from these policies are also the ones at greatest risk from eutrophication.

Water pollution and agriculture

Use of cost–benefit assessments for evaluation and design of N-policies for agriculture and aquatic ecology is still uncommon. In the case of the role of N_r for aquatic ecology major reasons appear to be the lack of a strong causality (e.g. in view of the role of phosphorus) and public awareness. The EU Water Framework Directive, in part dealing with trans-boundary transport of N_r in watersheds, does allow exemption to member states in the event that the costs of measures are disproportionate (e.g. to economic resources of a country or industry); either by making emission or water quality objectives less stringent or allowing extension of the time period to achieve the objectives.

For agriculture there is the complex balancing of the role of N_r for farm income, food security, environment and a level playing field on the world market. From a farmer's perspective it is profitable to add additional N fertilizer as it generates robust net revenues up to high input levels and increases the chance for return on investment in other production factors. The issues of food security, farm income and competitiveness have moved up the policy agenda since the food crisis in 2008, and the ongoing financial and economic crisis. There is a wide scope for increasing N_r-benefits in agriculture, particularly in NW Europe, but this requires a new kind of international cooperation to deal with the other issues at stake. The compelling obligation to feed Europe and the world could, for example, be combined with partial internalization of environmental costs by policies stimulating an increase of agricultural production in less productive regions of central, eastern and southern Europe. Competitiveness and reduction of environmental impacts could perhaps be combined by using the N-efficiency of agricultural production as a criterion for cross compliance. The N-efficiency would a priori benefit from strong EU wide regulations with respect to application of manure and discounting manure N within the application standards for total N.

Challenges for policy and research

Although integrated assessment of social cost and benefits of N in the environment is still under development it already provides guidance and useful insights, both in the domain of policy and science. In the domain of policy provisional results raise questions about the social benefits of present N-fertilization levels in agricultural production and future abatement options for air pollution by NO_x and NH_3. Another relevant conclusion appears to be that policy priorities for reduction of agricultural emissions of N_2O are not currently supported by the expected social benefits of this reduction, when valuing N_2O emission according to the CO_2-trading price.

Some more specific considerations for future N-policies are as follows.

- Increase the role of cost–benefit assessment as a supporting tool for policy evaluation and design: not as the economic truth about environmental policies but as a vehicle to increase transparency of policy decisions.
- Take into account co-benefits and side effects beyond the N-cycle when developing integrated N-policies.
- Consider to use N-efficiency of agricultural production as policy target and a criterion for cross compliance. Producing more food with less N_r is an important challenge for Europe and the world.
- Recognize the uncertainty in estimated abatement costs considering economy of scales and future technology improvements resulting in lower actual costs.
- Stricter regulation of manure application and improved N-efficiency, in view of the robust benefits for society.

Some specific issues for future research are as follows.

- Harmonizing methods to quantify and combine N_r-damage functions for human health, ecosystem health and climate change. The dose–response relations available for the role and share of N_r in ecosystem service provision are still very limited. Dose–response relations are available for the domain of human health but are subject to uncertainty (particularly for health impacts from airborne secondary ammonium and nitrate salt particles and for waterborne nitrate).
- Establish the long term effect of N-fertilization on crop yields. Although agronomic research into the yield response and economic benefit of N_r has a long history and a high standard we do not seem to understand the system well enough to find the key to higher N-efficiencies without affecting food security and economic vitality of the farming community.
- Quantify the role of airborne ammonium containing particles for health impacts. Present policies and data cause ammonia to be one of dominant cost items, while evidence for health risk of airborne ammonium and nitrate particles remains uncertain.

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Supplementary materials

Supplementary materials (as referenced in the chapter) are available online through both Cambridge University Press: www.cambridge.org/ena and the Nitrogen in Europe website: www.nine-esf.org/ena.

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