**Project Number 282910**

**ÉCLAIRE**

# Effects of Climate Change on Air Pollution Impacts and Response Strategies for European Ecosystems

**Seventh Framework Programme**

**Theme: Environment**

# D18.3 Elaboration of the Modelling Approach for Benefits Analysis, Including Illustrative Examples

Due date of deliverable: **01/05/2013**

Actual submission date: **23/07/2015**

Start Date of Project: **01/10/2011**

Duration: **48 months**

Organisation name of lead contractor for this deliverable :
**EMRC**

| Project co-funded by the European Commission within the Seventh Framework Programme | | |
|---|---|---|
| **Dissemination Level** | | |
| **PU** | Public | ☒ |

| | | |
|---|---|---|
| **PP** | Restricted to other programme participants (including the Commission Services) | ☐ |
| **RE** | Restricted to a group specified by the consortium (including the Commission Services) | ☐ |
| **CO** | Confidential, only for members of the consortium (including the Commission Services) | ☐ |

## 1. Executive Summary

The effects for assessment of this work package are to assess the economic value of impacts of atmospheric nitrogen and ozone on:

1. Terrestrial biodiversity
2. Crop production
3. Forest production
4. Carbon sequestration

These effects were selected as being particularly relevant to the pollutants of interest to ECLAIRE, and had the potential for quantification drawing on the outputs of other components of the study. It is noted that the methods and outputs of the analysis need to be in a form that can be integrated with cost-benefit analysis of European air quality policies.

This deliverable provides estimates of damage to biodiversity from deposition of nitrogen, in particular. Three methods for quantification of this important endpoint are considered, each adopting a different perspective:

- Public 'willingness to pay' for protection of biodiversity
- The costs of repairing damage
- Inferred preference for environmental protection drawing on the implications of the Habitats and Birds Directives.

It also provides estimates of damage to agricultural crop production through exposure to ozone. The valuation methods for forestry production and carbon sequestration are also reported.

Analysis has focused on scenarios used in the appraisal of the Clean Air Policy Package presented by the European Commission in December 2013. Special attention is paid to the Current Legislation (CLE) and Maximum Technically Feasible Reduction (MTFR) scenarios. The final analysis will be made using the ECLAIRE scenarios.

There is order of magnitude consistency in the results for biodiversity change generated by the three options identified above, with benefits of more advanced pollution controls in the range €800 million to €10 billion annually, though they imply that the benefits of protecting biodiversity are small compared to the benefits to health. For crops, benefits in the order of €2 billion annually will add to this total.

Consideration needs now to be given to how these results may be refined and then how they may be used to influence policy. Whilst health benefits will continue to dominate the analysis, the requirements of existing EU legislation on ecosystem protection are clear, and to achieve these requirements there will need to be further and substantial reductions in emissions.

## 2. Objectives:

Demonstration of approaches for assessment of change in the value of ecosystem services across policy relevant scenarios at the EU level.

## 3. Activities:

At a workshop held at RIVM in December 2013 it was decided that the key effects for quantification and valuation in work package 18 of the ECLAIRE Project related to impacts on biodiversity. Further effects on greenhouse gas balance, and impacts on forestry and agriculture are also being quantified under this work package, though these are conceptually far simpler to deal with as associated impacts can either be valued through existing markets (for agriculture and forestry) or using established costs per unit emission (for greenhouse gases).

It was accepted at the workshop that valuation of biodiversity is not straightforward, and that there is only a limited amount of information available that is relevant. For this reason it was decided to consider three alternative methods:

1. Application of available estimates of stated public willingness to pay (<u>WTP</u>) for protection of biodiversity. This approach has been applied at a full European scale. In areas where better data area available an approach that provides more explicit linkage of willingness to pay values to changes in species richness is also considered. Consideration is also given here to different factors that need to be taken into account to differentiate WTP across the EU, when relevant.

2. Adoption of <u>repair costs</u> as a proxy for WTP for protection of biodiversity

3. Assessment of inferred (revealed) preference with respect to emissions of air pollutants of policy makers when adopting environmental protection legislation (<u>regulatory revealed preference</u>). Given the extensive exceedance of the critical load for eutrophication and the requirements of the Habitats and Birds Directives, this implies a valuation at least equivalent to the costs of the MTFR (Maximum Technically Feasible Reduction) scenario.

Of the three methods the first is conceptually most robust. However, being based on a limited data set the other studies are considered useful for cross-checking.

Analysis is also provided for the quantification of damage from ozone effects on agricultural crops. Further information is provided relevant to valuation of ozone damage to forest, and effects of nitrogen and ozone on carbon sequestration.

Analysis is undertaken using scenarios considered for the European Commission's Clean Air Policy Package of 2013, considering a shift from the Current Legislation (CLE) scenario to the MTFR scenario. As noted elsewhere, these scenarios are used here purely for the purpose of illustration, to demonstrate the methods developed in the study.

## 4. Results:

Results for biodiversity effects are summarised in Table (i). For the Willingness to Pay (WTP) approach consideration was given to adjustment of WTP values according to several factors:

- Variation in income
- Variation in environmental concern as reflected by the Eurobarometer survey (Special Eurobarometer 365)
- Government expenditure on environmental protection

The second and third of these options were problematic for various reasons. In part, they were not sufficiently disaggregated to the levels of damage present across the region, nor to the specific issue of air pollution damage to ecosystems. For Eurobarometer the vast majority of people (95%) considered environmental protection to be important to some degree, with the effect that there was very little differentiation across countries (this strength of view is worth noting). With respect to government expenditure there were several problems also, particularly inconsistency between countries with respect to what was and was not included in the totals. Adjustment by income was thus considered to be the best option, though it, too, is not perfect.

**Table (i) Summary of results at EU28 level from the use of different methods for assessing the biodiversity benefit of moving between the CLE and MTFR scenarios in 2025 and 2030. Units €million/year**

| **2025** | **CLE** | **MTFR** | **Benefit of change** |
|---|---:|---:|---:|
| WTP | 3,175 – 9,525 | 2,294 – 6,822 | 881 – 2,643 |
| WTP with income adjustment | 2,678 – 8,034 | 1,856 – 5,568 | 822 – 2,466 |
| Repair cost | 9,096 | 6,361 | 2,735 |
| Regulatory revealed preference | 54,754 | 63,985 | 9,231 |
| **2030** | **CLE** | **MTFR** | **Benefit of change** |
| WTP | 3,116 – 9,347 | 2,211 – 6,633 | 905 – 2,714 |
| WTP with income adjustment | 2,621 – 7,863 | 1,781 – 5,343 | 840 – 2,520 |
| Repair cost | 8,745 | 5,999 | 2,746 |
| Regulatory revealed preference | 61,985 | 72,597 | 10,612 |

The results of Table (i) show roughly an order of magnitude variation in the results of the different methods, though there is much more consistency in the WTP and repair cost methods than with the regulatory revealed preference approach. At the European level income adjustment makes little difference to the WTP results, though it is more pronounced at the national level.

The quantified benefits of reducing emissions are noted to be small relative to the effects on health. However, the quantified effects include only one aspect of impact to ecosystems, the valuation of biodiversity. Further analysis presented here indicates that inclusion of impacts to crop production would add a further

€2billion to benefit estimates of moving from current legislation to the MTFR scenario.

Consideration needs now to be given to how these results may be used to influence policy. Whilst health benefits will dominate the analysis, the requirements of existing EU legislation are clear, and to achieve these requirements there will need to be further and substantial reductions in emissions.

## 5. Milestones achieved:

MS80: First complete set of scenario results.

## 6. Deviations and reasons:

The deliverable was submitted later than originally planned, but this has had no effect on other components as the benefits analysis is the final stage of the assessment. The additional time has allowed integration of additional methods and results, providing a better basis for discussion of the integration of ecosystem impacts with the broader benefits assessment of air pollution policies in Europe.

## 7. Publications:

None for this report, as it concerns illustrative application of methods.

## 8. Meetings:

The development of this report has been informed through a series of meetings, in particular:

- TFIAM (Task Force on Integrated Assessment Modelling)/NEBEI workshop, Zagreb, October 2013
- Workshop at RIVM, Netherlands, December 18 2013
- 4th Annual ECLAIRE Congress, Budapest, September 2014.

## 9. List of Documents/Annexes:

Elaboration of the Modelling Approach for Benefits Analysis for Biodiversity, Deliverable 18.3 Version 2. Mike Holland (EMRC), Rob Maas (RIVM), Laurence Jones and Gina Mills (CEH).

# Elaboration of the modelling approach for benefits analysis

# ECLAIRE Project, Work package 18

# Deliverable 18.3 Version 2

**Mike Holland (EMRC), Rob Maas (RIVM), Laurence Jones and Gina Mills (CEH)**

**With contributions also from IIASA, Aarhus University, IVL, and Stockholm Environment Institute**

**July 2015**

## Summary

The assessment of the impacts of ozone and nitrogen deposition on ecosystem services needs to account for a diverse range of impacts, from those on food production and carbon sequestration to the appreciation of biodiversity. The effects of ground level ozone on ecosystems are negative: the more ozone that plants are exposed to the worse they grow, the less food is produced, less carbon sequestered and so on. It is, however, noted that a clear relationship between ozone and biodiversity exposure has not been established in ECLAIRE. Nitrogen being an essential nutrient, however, presents a different picture. Some effects are negative, some are positive, at least in the short term. Plants adapted to live with limited nutrient availability will be outcompeted by grasses and other plants with higher nitrogen requirements when there is significant deposition of N from the atmosphere. This will lead to a reduction in biodiversity. On the other hand, forest growth will initially increase when more nitrogen enters the ecosystem, increasing productivity and carbon assimilation. However, as time goes on, forest ecosystems may become nitrogen saturated and not benefit from further deposition. Further to this, the addition of nitrogen may generate nutrient imbalances, for example through acceleration of the cycling of the base cations required to keep up with the growth induced by extra nitrogen, leading eventually to reduced growth and reduced carbon storage.

The nitrogen problem is exacerbated by our ability to quantify one side of the equation, the positive effects in the short term on production, more easily than those on the other side of the equation, the negative effects on biodiversity and on the sustainability of production into the future.

At a workshop held at RIVM in December 2013 it was decided that the key effects for quantification and valuation in work package 18 of the ECLAIRE Project related to impacts on biodiversity.

It was accepted at the workshop that valuation of biodiversity is not straightforward, and that there is only a limited amount of information available that is relevant. For this reason it was decided to consider three alternative methods:

1. Application of available estimates of stated public willingness to pay (WTP) for protection of biodiversity

2. Adoption of repair costs as a proxy for WTP for protection of biodiversity

3. Assessment of inferred (revealed) preference with respect to emissions of air pollutants of policy makers when adopting environmental protection legislation (regulatory revealed preference)

Of the three methods the first is conceptually most robust. However, being based on a limited data set (referenced to a series of key UK studies by Christie et al) the other approaches are considered useful at least for cross-checking.

Analysis is undertaken using scenarios considered for the European Commission's Clean Air Policy Package of 2013, considering a shift from the Current Legislation (CLE) scenario to the Maximum Technically Feasible Reduction (MTFR) scenario. Illustrative results considering effects of both NOx and $NH_3$ for the three methods are as follows:

**Summary of results at EU28 level from the use of different methods for assessing the biodiversity benefit of moving between the CLE and MTFR scenarios in 2025 and 2030. Units €million/year**

| **2025** | **CLE** | **MTFR** | **Benefit of change** |
|---|---|---|---|
| WTP | 3,175 – 9,525 | 2,294 – 6,822 | 881 – 2,643 |
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| Repair cost | 8,745 | 5,999 | 2,746 |
| Regulatory revealed preference | 61,985 | 72,597 | 10,612 |

**Notes:**

- The ranges for WTP and WTP with income adjustment are based on ranges from the Christie et al paper linked to protection of 'non-charismatic' species. Christie includes other ecosystem services that could be added to the estimates shown here. WTP with income adjustment as presented here uses an elasticity of 1.5, though results based on elasticities of 0 and 1 are also presented in the report.
- Results for the regulatory revealed preference method take the change in the cost of pollution abatement as a proxy for the benefits of control, hence the estimate for MTFR is higher than for CLE, unlike the other methods.

The variation in the benefit estimates is around one order of magnitude, when including the regulatory revealed preference approach. The restoration cost approach provides very similar estimates to the upper bound of the WTP range. The inclusion of additional ecosystem services would clearly increase in the WTP estimates.

The table implies that the effect of income adjustment is small. However, this is a function of the baseline value being for the UK which has broadly average income per capita compared to the EU28, hence the adjusted value for the EU28 is little different to the unadjusted. However, results in the main body of this paper show more significant variation between countries. The need to apply income adjustment would be linked to the scope of analysis (e.g. whether conducted for the European Commission or UN/ECE, or for individual countries).

Results imply that the benefits of biodiversity improvement are small relative to those quantified elsewhere for health impacts. However, as noted, there is some bias to underestimation, particularly through the limitation of the ecosystem services included for 'biodiversity'. Also, the true point of comparison is not with other benefits of possible interventions, but with the costs of intervention. These are of course reflected here in the results for the regulatory revealed preference approach. Whilst the results based on regulatory revealed preference when moving from CLE to MTFR are significantly larger than those from either the stated WTP or repair cost approaches, it is to be remembered that the cost curve is strongly non-linear (see Figure 2 in the main text). For a significant part of the range between CLE and MTFR it is possible that ecosystem benefits could outweigh costs without factoring in health and other issues. This then raises the need to consider the extent to which a strategy based on health protection would differ to one based on ecological protection.

A number of options for refinement of the methods are identified in the text. Some can easily be implemented and will be further explored.

In addition to demonstrating the use of these methods this report also reviews the strengths and weaknesses of each approach. The preferred approach of the three is the use of stated preference WTP. The next phase of the work is thus to investigate how the outputs of the stated preference literature can better be linked to other analysis being performed in ECLAIRE. In this report a simple approach has been taken based on the change in area subject to critical loads exceedance. However, discussions are underway to link to other metrics, including indices of biodiversity. Progress has been made in this direction in work being conducted in the UK.

Analysis is also presented of impacts of ozone exposure on crop production in Europe. This adopts the now-preferred dose-based functions to the extent that they are available (for wheat, potato and tomato). However, it then extends the analysis to all European crop production. Results indicate that moving from current legislation to the MTFR scenario would generate benefits of €2 billion per year. Impacts are dominated by effects on wheat, 30% of the total. Together, wheat, potato and tomato, the three crops for which dose-based functions are available account for 40% of damage.

# Contents

# 1 Introduction

## 1.1 ECLAIRE objectives

The ECLAIRE Project (Effects of Climate Change on Air Pollution Impacts and Response Strategies for European Ecosystems), funded by European Commission DG Research under the 7$^{th}$ Framework Programme, has the following broad objectives of relevance to the present report:

- To investigate the ways in which climate change alters the threat of air pollution (NOx, $NH_3$ and ozone) on European land ecosystems including soils. This includes the development of response relationships based on field observation, experimental data and modelling.

- To quantify how climate change alters ecosystem vulnerability to tropospheric $O_3$ and N deposition, including interaction with increased $CO_2$. Combined with special topics on interactions with N form (wet/dry, NHx/NOy), aerosol-exacerbated drought stress and BVOC self-protection of $O_3$ effects, novel threshold and dose-response approaches will be developed.

- To estimate interactions and feedbacks on plant and soil carbon stocks, greenhouse gas balance and plant species change.

- To apply the new risk assessment chain at the European scale, to assess how projected climate change will alter damage estimates, in part through economic valuation of ecosystem services. Improved integrated assessment modelling will allow cost-benefit analysis to better inform future mitigation and adaptation strategies on air pollution and climate change.

## 1.2 Work Package 18 objectives

Within the Project, Work Package 18 is designed to derive economic impacts and valuation of changes[1] in ecosystem services through the following objectives:

- To link the concept of ecosystem services with existing mapping of European ecosystems and pollutant impacts.

- To characterise the links between pollutant exposure, impact and value to permit quantification of pollutant damage.

[1] The focus on change is important: the analysis does not seek to ascribe value to the totality of European environmental service, but to the change arising from moving between different policy scenarios.

- To assess change in the value of ecosystem services across different scenarios using a marginal approach to the extent possible.

- To prioritise gaps in the existing knowledge base such that further research can be targeted on the parameters likely to have the greatest economic impact.

A constraint on the analysis is that ECLAIRE does not include original valuation work, but is instead focused on the application of available valuation studies to the ECLAIRE outputs.

The ecosystem services concept has been particularly developed through the TEEB (The Economics of Ecosystems and Biodiversity) Initiative[2]. TEEB highlights that all human societies and communities recognise value in ecosystems, landscapes, species and other aspects of biodiversity, and this alone may be sufficient to ensure conservation and sustainable use. Acknowledging and understanding the nature of 'value' in this sense provides a framework for further analysis where this is required. Demonstrating value in economic terms can assist policy makers and others in reaching decisions that seek to consider the full costs and benefits of an ecosystem rather than just those costs or values that enter the markets in the form of private goods. It is acknowledged that a 'full' accounting of costs and benefits is often not possible beyond the conceptual stage, but, equally, even this provides a higher level of information than has typically been available in the past. A third element of TEEB is the development of mechanisms for factoring these values, quantified or unquantified, into policy through, on the one hand, incentivisation of practices that protect the environment, and on the other hand, penalties for actions that would cause damage.

## 1.3 The objectives of this report

At a workshop of this Work Package held at RIVM in December 2013, it was concluded to adopt a framework for quantification of damage to biodiversity investigating the use of three different approaches. This recognises that no single method is likely to generate perfect results at the present time, but that each may provide some insight.

The first of the three methods is based on available information from stated preference studies regarding the 'willingness to pay' (WTP) for protection of biodiversity. The second is based on use of repair costs, and the third on the inferred costs of environmental policies to the extent that these have not been made explicit, both of these being 'revealed preference' approaches.

It was also decided that the work should give particular attention to the Natura 2000 network. This is established through the Special Protection Areas of the

[2] http://www.teebweb.org/

Birds Directive[3] and the Special Areas of Conservation of the Habitats Directive[4]. These impose a legal responsibility on Member States to:

- "*preserve, maintain or re-establish a sufficient area of habitats for all of the birds [*referred to in the Directive]" [Birds Directive], and
- "*maintain or restore, at favourable conservation status, natural habitats and species of wild fauna and flora of Community interest*." [Habitats Directive]

The reasons for a particular focus on biodiversity in this report were as follows:

1. Existing assessment of the monetised benefits for European air pollution policies focuses almost exclusively on health impacts (Holland, 2014). Impacts on biodiversity were one of the original concerns that led to the development of integrated European policies on air pollution. Information on the risk to biodiversity from eutrophication indicates that there is still a long way to go before it may be considered that there is a good level of protection from airborne pollutants.
2. Jones et al (2013) demonstrated the potential for quantification at the level of the UK, drawing on research by Christie (2006, 2011). Associated monetary values of effects on biodiversity were a significant part of the total quantified by Jones et al.

An illustrative quantification of impacts to crops and forests is also applied.

The scenarios considered here are not the final ECLAIRE scenarios. Quantification for these will be performed in a further report.

## 1.4 The rationale for valuation

The objective of economic valuation as applied here is to describe public preference for the efficient allocation of scarce resources (money). Results do not seek to provide insight about the 'fundamental value of nature', but instead deal with public perception of how the public would like money to be spent.

Following on from this, some readers may question the very notion that it is appropriate to define monetary values for natural assets. This argument overlooks the fact that decisions on the value of natural assets are taken routinely by policy makers, but those valuations are implicit to the decisions reached. Few would argue against the interests of consistency in policy making being better served by making valuations explicit, as part of the full rationale for any decision.

A particular complication for valuation of impacts to biodiversity is the time dimension. For impacts to health and production from agriculture and forestry this is less important. Some of the impacts are clearly annual (e.g. short term effects of exposure to pollutants on health, or effects on crop production) whilst others can be managed, for example by adjusting harvesting regimes in forests. The longer-term effects on health are quantifiable and limited by life expectancy,

[3] http://eur-lex.europa.eu/legal-content/EN/ALL/?uri=CELEX:31979L0409
[4] http://eur-lex.europa.eu/legal-content/EN/TXT/PDF/?uri=CELEX:31992L0043&from=EN

with effects accruing over a period of perhaps 20 years prior to death (this being the longest lag considered by various health analysts, considered appropriate, for example, where air pollution initiates lung cancer). Effects on ecosystems, in contrast may persist over decades, centuries, or be permanent. This issue is returned to in the discussion.

The terms 'willingness to pay' (WTP) and 'willingness to accept' (WTA) are used at various times in the present report. They are both standard economic concepts: for any economic transaction to take place one party has to be willing to pay an amount that the other will accept for a good. The concept applies not only to material goods. For example, when paying for additional safety equipment in a car, one is not paying for the safety equipment per se, but for the reduction in risk of death or injury that it provides. WTP applies when a risk or impact is reduced, WTA applies when a risk or impact increases.

## 1.5 Accounting Framework for ecosystem services

### 1.5.1 Prioritisation of effects for analysis

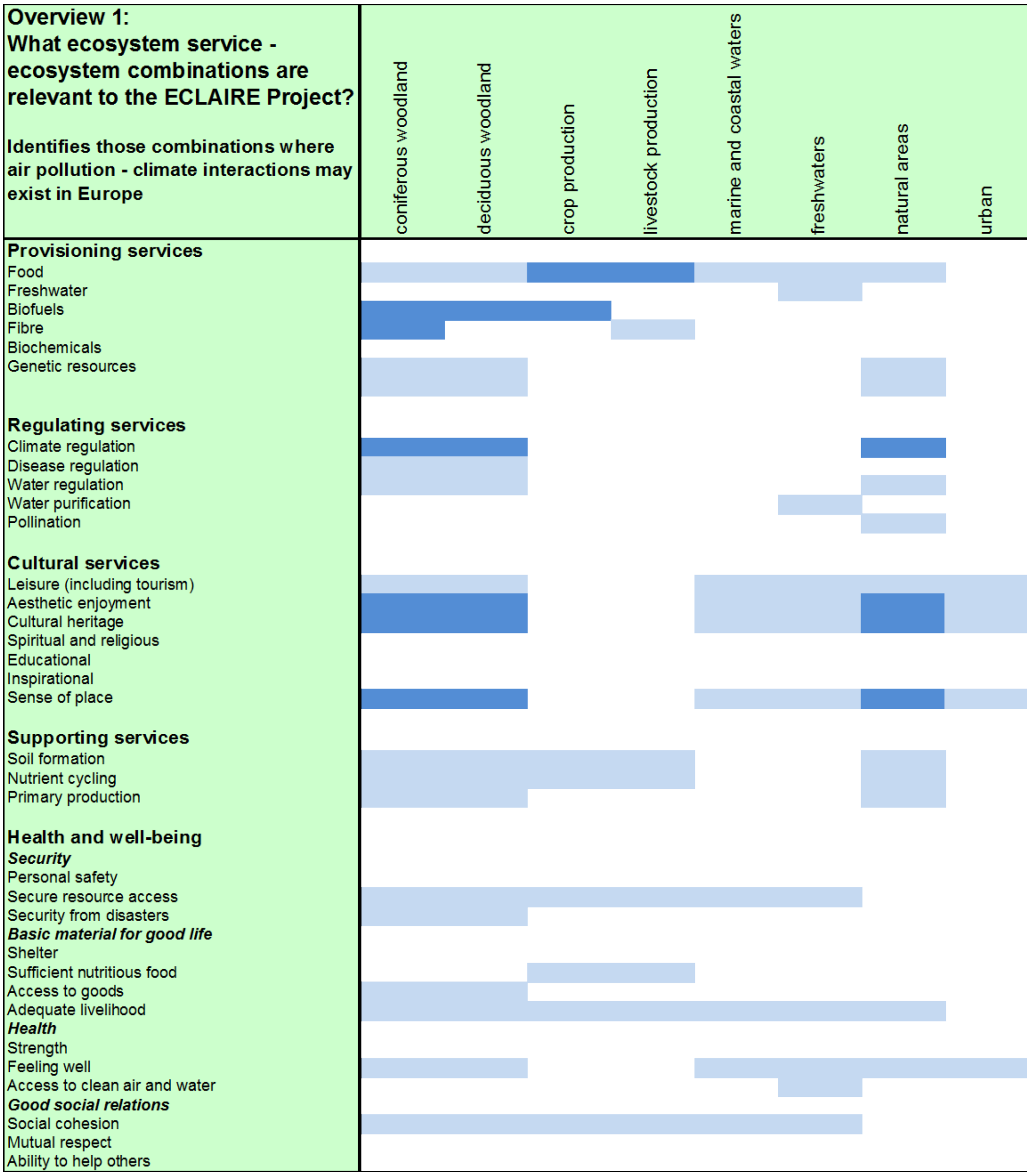
In order to understand how complete the assessment of impacts and their economic valuation is, an 'accounting framework' has been generated as part of this work package. The accounting framework shows how impacts, ecosystem services and different types of ecosystem are linked. An overview of linkages is presented in Figure 1. Cells highlighted light blue are those where there is potential linkage to the objectives of ECLAIRE, those highlighted in dark blue have been selected as priorities for the economic assessment (further information on the prioritisation process is provided in other deliverables under this work package). The prioritisation is based on the following criteria:

1. Relevance to the objectives of ECLAIRE
2. Relevance to other Work Packages in ECLAIRE
3. Likely availability of data on stock at risk, response and valuation
4. Perceived importance

Figure 1 takes a deliberately broad definition of ecosystem services, including health and well being, as there are issues here that will not be covered elsewhere in existing valuation frameworks for air pollution policy analysis. The listing of ecosystem types (coniferous woodland, deciduous woodland, crop production, livestock production, marine and coastal waters, freshwaters, natural areas and urban) is intended to be aligned with the structure of analysis and the types of effect that may be quantified (or not) through ECLAIRE. The somewhat loose term 'natural areas; is intended to cover grassland, moors, bogs, etc. that are not covered elsewhere.

The prioritisation process clearly needs to take the feasibility of quantification into account. Recent work by Jones et al (2013) provides results across a number of ecosystem services for the UK, reflecting the costs and benefits of declining nitrogen deposition. Two provisioning services were covered (timber and livestock), one regulating service (GHG balance) and two cultural services (recreational fishing and appreciation of biodiversity). Across these effects there was a mix of losses and gains. Provisioning services were forecast to have experienced a loss against reduced N deposition in the UK in recent years, through reduction of N fertilisation, whilst cultural services benefited. The effect on GHG emissions varied by gas, though dominated by reduced $CO_2$ sequestration, reflecting reduced N fertilisation. Results are dependent on the precise scenario under investigation, and also assumptions of long term impacts – of particular note is the question of whether in the long-term elevated levels of N deposition are sustainable – a question that is receiving much debate through ECLAIRE.

## Figure 1. Links between ecosystem services and ecosystem types.



### 1.5.2 Underlying information in the Accounting Framework

Additional information is then provided within the accounting framework to identify the impacts covered under each cell of Figure 1. For example, there are numerous effects of the ECLAIRE pollutants on crop production:

1. Direct effects of N, ozone and $CO_2$ on crop production, forest growth and carbon sequestration
2. Direct effects of N, ozone and $CO_2$ on the nutritional value of crops
3. Indirect effects of N, ozone and $CO_2$ on the performance of insect pests and pathogens [though noting that there has been little research in this area in the last 20 years].
4. Direct injury to leaf crops from exposure to high levels of ozone in sensitive periods

Doubtless other effects can be added to this list. By bringing together this information it is possible to gain perspective on how complete the analysis is, understanding precisely what is included and excluded. This may lead to additional effects being brought into the assessment. Without the framework it would be easy to believe that analysis is far more complete than it really is.

The economic analysis can also be further substantiated with additional contextual information that provides illustration and adds meaning to impact estimates. One important reason for wanting to do this arises from the nature of pan-European analysis. Multiplying a small impact by 530 million people can easily generate a substantial estimate of damage in the order of billions of euro. Contextual information can assist in helping stakeholders reach conclusions as to the likelihood that estimated damage is a reasonable reflection of reality.

Contextual information can take many forms, for example, maps showing the exceedance of critical loads across Europe, photographs of damage to crops, graphs showing response functions, etc. An excellent initiative has been launched by the ICP Vegetation, providing a mobile phone application to photograph and collate ozone damage across Europe[5].

[5] http://icpvegetation.ceh.ac.uk/record/mobile-app-ozone-injury

# 2 Previous economic analysis of air pollution in Europe

This chapter provides an overview of the most relevant studies for the purpose of this report. For this reason this chapter does not provide detailed discussion of either TEEB (The Economics of Ecosystems and Biodiversity) or of MAES (Mapping and Assessment of Ecosystems and their Services), but notes here their general relevance to the ECLAIRE work[6].

## 2.1 European cost-benefit analysis (CBA) to support the development of air pollution policies

This section describes the overall framework for CBA in relation to air pollution policy assessment in Europe. It applies equally to work done for the European Commission and the UNECE Convention on Long Range Transboundary Air Pollution.

CBA has been applied to European air pollution policy development since the mid 1990s, during the development of the EU's Acidification Strategy (Holland and Krewitt, 1996). The costs of meeting pre-determined objectives for environmental quality were assessed using the RAINS model (the precursor to GAINS; IIASA, 1996). The monetised benefits analysis at that time focused on quantification of effects on health and building materials of reductions in emissions of $SO_2$, NOx, $NH_3$ and VOCs, largely linked to the formation of secondary inorganic aerosols and ozone, drawing on methods developed as part of the EC DG Research ExternE (Externalities of Energy) study. Ecosystem benefits were not described in economic terms, but using physical and chemical parameters (the area of land subject to critical loads exceedance and the accumulated exceedance of critical loads). The demonstration of significant health impacts was a surprise to many of those participating in discussions on the Strategy.

Analysis was subsequently undertaken for the first Gothenburg Protocol to the UN ECE Convention on Long Range Trans-boundary Air Pollution (CLRTAP) (IIASA, 1999a; Holland et al, 1999), the EU's Ozone Directive (AEA Technology, 1998) and the National Emission Ceilings Directive (IIASA, 1999b, AEA Technology, 1999) again using the RAINS model to assess costs and ExternE methods to quantify health benefits. Again, the objectives of the legislation were primarily focused on ecological protection against acidification, eutrophication and ozone. The benefits analysis was extended to include some additional ecological impacts. Effects of ozone on crop damage and forest growth were accepted for inclusion in the final report, but assessment of the benefits of protection for (semi-) natural ecosystems were not. The reasons for rejection were that available estimates from the economics literature either did not link to the metrics used to define ecological improvement, or considered unreasonable scenarios (such as the total elimination of acidifying emissions). The results of the CBA did not play a large role in the decision making process at international

[6] TEEB: http://www.teebweb.org/about/. MAES: http://biodiversity.europa.eu/maes.

level, being used mostly to affirm that a cost-benefit assessment had been performed and had generated a positive outcome for the policy proposals under consideration.

Analysis for the EU's Clean Air For Europe (CAFE) Programme in 2005 (AEA Technology, 2005) and for the revision of the Gothenburg Directive in 2011 (Holland et al, 2011) covered a similar range of impacts, mainly health with also some assessment of benefits through reduced ozone exposure of crops and reduced acidification and ozone damage to materials used in buildings and to rubber goods. Benefits to ecosystems were reported in physical/chemical terms. The CBA was again used to affirm that the benefits of the proposed actions would exceed the estimated costs of action.

The most recent air pollution policy analysis at the EU level was undertaken to inform the development of the Clean Air Policy Package, released on 18th December 2013 (IIASA, 2014; Holland, 2014)[7]. The scope of the work was similar to the earlier analysis, again lacking monetization of ecological benefits. A notable difference, however, was that marginal benefits were used to define the ambition level (**Figure 2**).

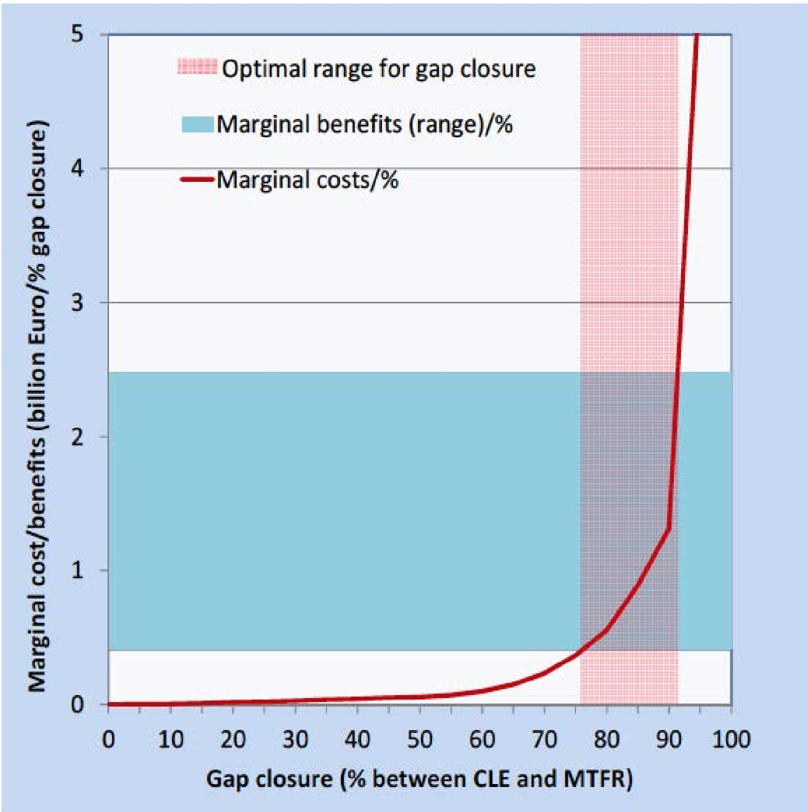



**Figure 2. Identifying an appropriate ambition level for the EU's Clean Air Policy Package (from IIASA, 2014)**

[7] For full details of the impact assessment of the Clean Air Policy Package see: http://ec.europa.eu/environment/air/clean_air_policy.htm

The figure shows on the x-axis the 'gap closure' between the baseline CLE (current legislation) scenario and the upper bound MTFR (maximum technically feasible reduction) scenario. The y-axis shows marginal costs or benefits in €billions per percentage point gap closure. The red line shows marginal costs, which increase as one moves towards the right hand side of the graph as successively more expensive/less cost-effective measures are adopted. The blue band shows the range for mortality benefits linked to a reduction in $PM_{2.5}$ exposure, the range corresponding to alternative valuations of the mortality impact. The marginal benefits line is horizontal because the health impact assessment uses a linear response function for mortality with no threshold. Benefits can be maximized by taking the analysis to the point at which marginal costs and benefits are equal, corresponding to a gap closure of between 76% and 91%. The policy objective was defined around the 76% gap closure level.

The approach taken is conservative for several reasons:

- It selects the point at which the lower (rather than the higher) estimate of mortality benefits is equal to the marginal costs.
- It only considers the mortality effects of fine particles, ignoring morbidity effects and all health effects of exposure to ozone
- It completely omits consideration of non-health impacts, including ecosystem benefits
- It takes the estimate of costs as given, without uncertainty.

Despite this in-built conservatism, the Clean Air Policy Package settled on a higher level of ambition in terms of % gap closure to MTFR than previous political discussions, suggesting a far more active role for the benefits analysis than in the earlier work.

Figure 3 shows the relative share of damage to health (as mortality and morbidity), crops and materials in 2030 across the EU from work on the Clean Air Policy Package. The analysis for crops and materials was performed using simplified damage per tonne factors, recognising that it would represent only a small part of the total. Considering ozone alone, it should be noted that crop damage accounts for 13% of total impact.

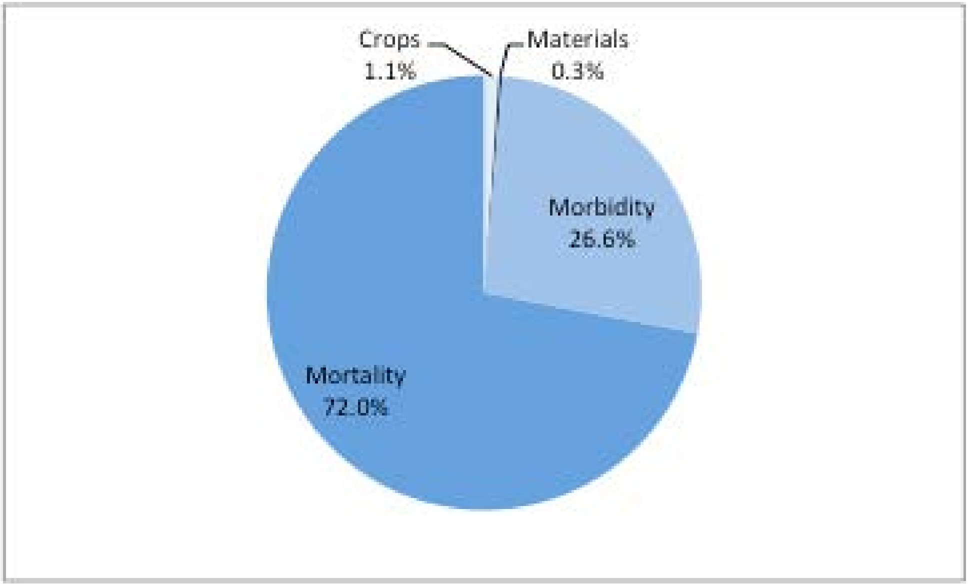



**Figure 3. Relative share of estimated damage from exposure to fine particles and ozone in the EU28 in 2030 (based on data in Holland, 2014).**

## 2.2 Crop damage assessments at the European scale

A number of crop damage assessments have been carried out at the European scale, principally through the ICP Vegetation under the UN/ECE Convention on Long-Range Transboundary Air Pollution. The most recent of these (Mills and Harmens, 2011) demonstrates the application of functions based on the phytotoxic ozone dose metric (POD), in contrast to earlier work that focused on use of concentration expressed as ozone concentration accumulated through the growing season at concentrations in excess of 40 ppb (AOT40). Results indicate losses of €3.2 billion and €1.0 billion for wheat and tomato respectively, in 2000. Action to reduce precursor emissions reduces this damage to €1.96 and €0.63 billion respectively, by 2020. POD relationships were not available at the time of writing for other crops, though a POD function is now available for potato.

## 2.3 Willingness to pay

### 2.3.1 Older studies

Markandya (in ExternE, 1995) reviewed available studies on ecosystem valuation and found none with relevance to the impacts of energy production or air pollution. Some were too species specific, some focused on very specific ecosystems that demonstrated the WTP method in relation to specific infrastructure projects but did not generate results of wider relevance. Pearce et al (2006) provides a review of the use of the ecosystem services approach and its relevance to environmental CBA, but at the time had little substantive and quantitative material to consider. The problems experienced have concerned, particularly:

- Incomplete knowledge of pollution impacts on ecosystems
- Lack of response functions for marginal analysis
- Lack of valuation studies on marginal change

Brink and van Grinsven (2011) reviewed the state of science in this area for the European Nitrogen Assessment. They noted that the TEEB-COPI study (Braat and ten Brink, 2008) listed only three studies that provide data to derive unit cost values for ecosystem services linked to N. A value of €2.2/kg N was given for 'Water purification and waste management' both for scrubland and grassland, and €25/kg NOx -N for 'Air quality maintenance'. Pretty et al. (2003) quantified costs of freshwater eutrophication in England and Wales. Brink and van Grinsven noted two problems with this study, that it mixed control and damage costs, and made no distinction between the effect of N and phosphorus. Söderqvist and Hasselström (2008) estimated WTP for a clean Baltic Sea, updating results from Contingency Valuation surveys in the 1990s. In this survey a random sample of respondents was questioned 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-load. Values ranged from €70 to 160 per household for the Eastern Baltic States with lower GDP, and between €500 and 800 per household for the Baltic States with high GDP. Gren et al (2008) report a range of unit damage costs of 12–24 euro per kg N based on Söderqvist and Hasselström (2008), using different discount rates.

For this reason alternative methods (as described below, repair costs and regulatory revealed preference) have also been considered by some authors – not regarding them as superior to a willingness to pay approach so much as accepting that the data necessary for a WTP assessment were unavailable and that ecosystem impacts should be factored into assessment.

### 2.3.2 Jones et al (2013)

Jones et al (2013) for UK government provides the most detailed and extensive of recent economic assessments of air pollution and ecosystems for the purposes of linking to the outputs of ECLAIRE activities. The pollution scenarios considered are specific to the UK, though this is not of great concern here, as we are here concerned with the approaches used, not so much existing results. Analysis demonstrates the application of the results of a study by Christie et al (2006, 2011) that considered the valuation of changes in biodiversity in the UK, work originally undertaken as input to the review of the UK's Biodiversity Action Plan (BAP).

Jones et al followed an impact pathway approach to assess impacts on biodiversity. In a first stage, response curves were generated to show %habitat damage against N deposition (Figure 4). This was valued using data selected from Christie et al (2011). Christie provides data for a number of ecosystem services (an illustrative set is provided in Table 1. The options investigated by Christie were WTP to ensure the then current BAP was implemented, and WTP for an enhanced BAP. Jones et al took the values for 'non-charismatic species' (£88/household/year for implementation of the current BAP, £44/household/year for the enhanced BAP) and scaled across the ranges for critical loads exceedance as shown in Figure 5[8].

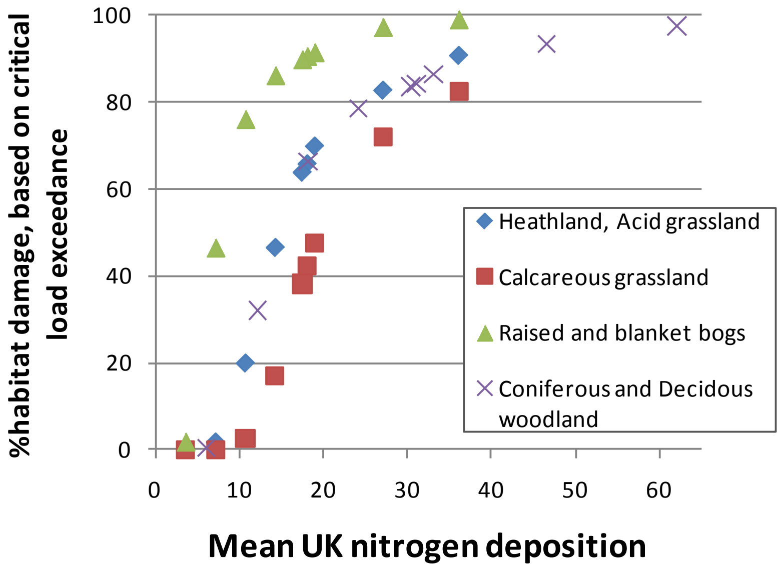



**Figure 4. Response curves for %habitat damage (based on critical load exceedance) against mean UK nitrogen deposition (kg N ha$^{-1}$ yr$^{-1}$).**

**Table 1. Estimated WTP for ecosystem services delivered by BAP habitats within the respondents' own regions of the UK (Christie et al, 2011)**

| Ecosystem service attribute | BAP scenario | UK (pooled model) (£ per household per year) |
|---|---|---|
| Wild food | Increased spend<br>Current spend | £34<br>£79 |
| Non food products | Increased spend<br>Current spend | £40<br>£53 |
| Climate regulation | Increased spend<br>Current spend | £98<br>£168 |
| Water regulation | Increased spend<br>Current spend | £80<br>£150 |
| Sense of place | Increased spend<br>Current spend | £89<br>£71 |

[8] For consistency with the original studies, we retain use of UK £ here. At the time of the study the exchange rate was €1.18/£.

| | | |
|---|---|---|
| Charismatic species | Increased spend<br>Current spend | £108<br>£115 |
| Non-charismatic species | Increased spend<br>Current spend | £44<br>£88 |

Source: Table 15; Christie et al. (2011).

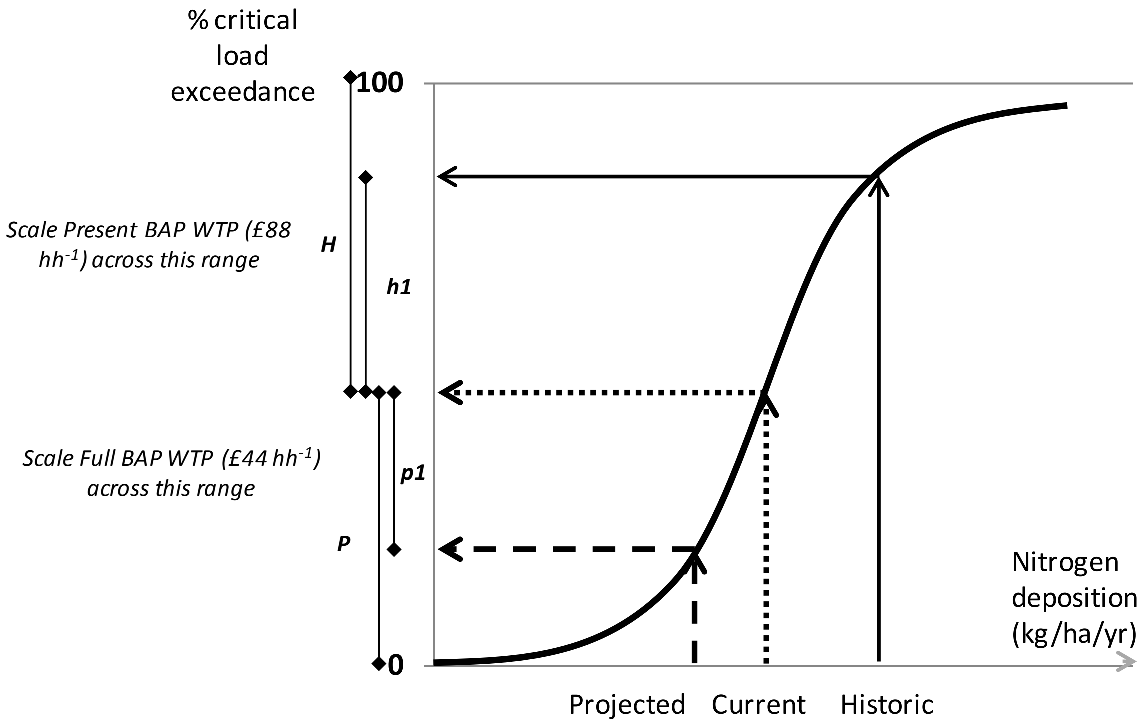



**Figure 5. Method for scaling WTP values across critical load exceedance ranges.**

**Notes:** Estimated WTP from Christie et al. (2010) was scaled linearly across the range of percentage damage as follows: For projected declines in N deposition, WTP of £44 per household was scaled by progress towards a state of zero exceedance from current levels of damage (£44 * p1 / P). For historical emissions where N deposition, and therefore habitat damage were higher in the past, the difference between current damage and 100% habitat damage was used (£88 * h1 / H).

The linkage of the response curve and valuation is a significant improvement on the other methods considered here.

Christie et al (2011) reports an expert weighting of the importance of different habitat types for the delivery of each ecosystem service. For the habitats of interest to Jones et al (considering that their analysis was concerned only with air pollution threats) this weighting combined to a proportion of 0.479 of the available weighting, and hence only this proportion of the total WTP of £44/£88 per household was applied. The use of habitat specific WTP values enabled separate calculation of % damage using habitat-specific critical loads data and WTP for each habitat, again, a significant advantage over the other methods considered.

At first sight, Jones' analysis might be thought of as assuming that recovery of ecosystems would occur without time lag, immediately following the decline in N deposition. This would clearly be unrealistic. However, lag effects are implicit to the valuation of Christie et al as some lag should have appeared implicit to respondents to the WTP survey, not least as it considered two scenarios of improvement (current and extended UK BAP).

Results from Jones et al are repeated in Table 2, demonstrating the range of ecosystems considered and the relative importance of damage to each under historic and projected emissions. [For the purpose of comparison with later results in this paper select the annualised estimates (EAV – Equivalent Annual Values) shown.]

**Table 2. Estimated value of air pollution impact on appreciation of biodiversity (present value and equivalent annual value) (£ million).**

| **Air pollution impact** | **Habitat** | **Scenario** | |
|---|---|---|---|
| | | ***Historical emissions (1987 – 2005) vs. baseline*** | ***Projected emissions (2005 - 2020) vs. baseline*** |
| Nitrogen impact on biodiversity | Woodland | PV: £143m<br>£20.7m to £266m<br>EAV: £10.8m<br>£1.8m to £20.5m | PV: £27.6m<br>£4.0m to £52.5m<br>EAV: £2.4m<br>£0.3m to £4.5m |
| | Heathland | PV: £362m<br>£51.7m to £671m<br>EAV: £27.4m<br>£4.0m to £50.3m | PV: £221m<br>£31.9m to £410m<br>EAV: £19.1m<br>£2.7m to £35.7m |
| | Acid grassland | PV: £55.2m<br>£7.9m to £102m<br>EAV: £4.2m<br>£0.6m to £7.8m | PV: £33.7m<br>£4.8m to £62.8m<br>EAV: £2.9m<br>£0.4m to £5.3m |
| | Calcareous grassland | PV: £333m<br>£48.1 to £1,016m<br>EAV: £25.3m<br>£3.6m to £47.0m | PV: £433m<br>£63.3m to £778m<br>EAV: £37.6m<br>£5.4m to £68.1m |
| | Bogs | PV: £263m<br>£37.6m to £810m<br>EAV: £19.9m<br>£2.8m to £36.1m | PV: £31.0m<br>£4.4m to £57.4m<br>EAV: £2.7m<br>£0.4m to £4.8m |
| | Total | PV: £1,156m<br>£167m to £2,141m<br>EAV: £87.7m<br>£13.1m to £163m | PV: £745m<br>£107m to £1, 357m<br>EAV: £64.7m<br>£9.2m to £121m |

Notes: PV = present value benefit, estimated applying Green Book guidance for discounting (HM Treasury, 2003); EAV: equivalent annual value. Figures provided are left in original currency, exchange rate for the year of survey approximately €1.18/£.

### 2.3.3 Variations on the scaling of WTP approaches to air pollution metrics

As noted above, Jones et al (2013) used WTP values from Christie et al (2011) to scale impact according to critical load exceedance (Figure 5). Subsequent work has focused on more explicit linkages of WTP values to changes in species richness.

Value transfer scaling according to changes in species richness can be based on another valuation study by Christie et al. (2012) which looked at WTP to manage UK protected areas (Sites of Special Scientific Interest – SSSIs) under different funding scenarios. They conducted a choice experiment for selected attributes related to ecosystem services, including changes in plant and animal species. These species attributes were framed explicitly in terms of a percentage change in the populations or range of threatened species (Table 3). The clear description of the choice outcomes allows direct linkages across to measures of damage due to air pollution. As a result, it is possible to directly scale changes in species richness to WTP estimates, as illustrated in Figure 6. Using this approach, a percentage change in species richness, calculated from absolute or relative values species richness changing in response to N deposition is scaled according to the 25% increase (assumed direction of response to falling N deposition levels) or 50% decrease (assumed direction of response to increasing N deposition levels) specified in the WTP study.

**Table 3. Components of the choice experiment by Christie et al. (2012) framing the consequences for non-charismatic and for charismatic species under different funding scenarios.**

| Attribute | Increase SSSI funding | Maintain SSSI funding | Remove funding |
|---|---|---|---|
| Non-charismatic species | 25% increase in the population or range of threatened species | No change in the population or range of threatened species | 50% decline in the population or range of threatened species |
| Charismatic species | 20% increase in the population or range of threatened species | No change in the population or range of threatened species | 55% decline in the population or range of threatened species |

One approach for modelling changes in biodiversity in response to changing air pollution is that of 'habitat suitability' broadly defined as the levels of nitrogen and sulphur pollution which would support a defined list of typical or appropriate indicator species for a habitat type. In principle, a response function could also be developed to provide read across from WTP studies such as Christie et al. (2012) to changes in the habitat suitability metric with changes in nitrogen or sulphur deposition.

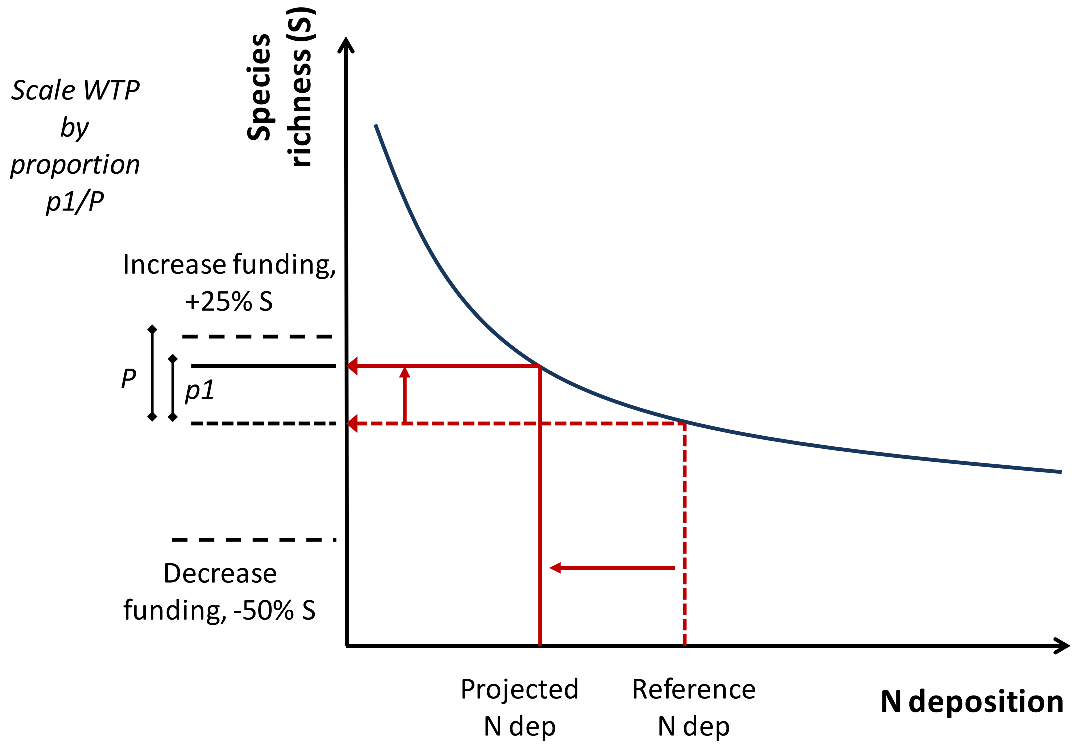



**Figure 6. Scaling WTP for changes in species richness according to funding scenarios from Christie et al. (2012).**

## 2.4 Restoration cost approach

Recognising that the omission of damage to ecosystems was a major gap in the ExternE toolset, Ott et al (2006, under the NEEDS Project) presented an approach for assessing biodiversity losses due to energy production, including effects of $SO_2$, NOx and $NH_3$ based on the use of repair costs. The use of repair costs has a distinct advantage over other methods in being able to utilise cost data from the real market of ecosystem restoration, though it is not without its problems. Analysis was performed in two stages:

1. Assessment of the 'potentially disappeared fraction' (PDF) of species due to pollutant deposition, drawing on previous studies by Eco-indicator (1999) and Koellner (2001).

2. Valuation of estimated PDF changes using a restoration cost approach.

Ott describes the process for determination of the baseline state for a particular land-use as follows:
*The PDF of vascular plant species is expressed as the relative difference between the number of species S on the reference conditions and the conditions created by the conversion, or maintained by the occupation. Basing on these data, PDF was calculated as follows:*

$PDF = 1 – S_{use} / S_{reference}$

*where $S_{use}$ is the species (richness) number of an occupied or converted land use type and $S_{reference}$ is the average species number in the reference area type. The species number of a specific land use type is standardised for 1 $m^2$. This absolute species number is transformed into a relative number using the regional species richness of the Swiss Lowlands* [40 species per $m^2$] *as a reference.*

PDF values are then given for a series of different CORINE land types.

For the impact assessment, modelling of the change in PDF was performed by first estimating the Probability of Occurrence (PO) for different plant species in different ecosystem types under different levels of acidification and eutrophication. The PDF is then calculated as

PDF = 1 - PO

The PDF can be interpreted as the fraction of species that has a high probability of no occurrence in a region due to unfavourable conditions caused by acidification and eutrophication.

Dispersion modelling to describe NOx, $SO_2$ and $NH_3$ deposition was undertaken using a Dutch model (Natuurplanner), and the changes in deposition were then translated into changes in the PDF drawing on information from Eco-indicator 99 (1999) which contains information for more than 40 types of ecosystems. For the purpose of the modelling, Ott et al considered that a species would be at significant risk ('stressed') if its probability of occurrence was less than a threshold of 2.5%. The number of stressed target species was counted and the results aggregated for the total natural area in the Netherlands, resulting in a percentage of threatened species caused by a specific level of deposition.

In the case of valuing biodiversity losses due to deposition of airborne emissions, the average costs of restoration of more or less natural areas (for which a very broad definition was applied: essentially any area that is not urban) to land use categories with high biodiversity were considered. Cost estimates were based on the cheapest habitat restoration choices available to bring about a significant improvement in biodiversity through the PDF concept. Using German data on restoration costs a marginal cost of 0.49 €/(PDF*$m^2$) was calculated. [The mix of Dutch and German data, and their extrapolation to the European scale, highlights the limited availability of information for this analysis].

Ott et al cite the following assumptions used in the calculation of external costs per kg of air pollutant for different countries:

1. The PDF change per mass of pollutant (PDF/kg deposition per $m^2$) as derived for the Netherlands is the same for all European countries.
2. The marginal costs of 0.49 €/(PDF*$m^2$) calculated for Germany need only be corrected by purchasing power (PPS) to be valid for other countries as well.

3. Degradation (a change in PDF) only takes place on natural land. According to ten Brink et al (2000), natural land encompasses all CORINE land use classes except the classes 1 (artificial areas) and 2 (agricultural areas) (for a CORINE list, see Koellner 2001).
4. The background level of acidification and eutrophication of the respective country influences the impact of additional depositions on biodiversity and hence the resulting external costs.

Ott et al then presented results in terms of external costs per unit of PDF change and per kg deposition of sulphur and nitrogen. It is understood that these results account for the fraction of natural areas within each country, and hence that the total damage per country can be estimated by multiplying the figures shown by total national deposition (see Table 15 of Ott et al). Results for NOx and $NH_3$ are shown in Table 4. Finally, they sought to validate their results using information from available WTP studies.

**Table 4. External costs per kg deposition for NOx and $NH_3$, PPP adjusted, €/kg for 2004.**

| | NOx | $NH_3$ |
|---|---|---|
| EU25 | 0.75 | 1.88 |
| Austria | 1.51 | 3.91 |
| Belgium | 0.96 | 2.49 |
| Bulgaria | 0.06 | 0.35 |
| Croatia | 0.65 | 1.55 |
| Cyprus | 0.01 | 0.08 |
| Czech Republic | 0.54 | 1.41 |
| Denmark | 0.40 | 1.13 |
| Estonia | 0.50 | 2.16 |
| Finland | 1.36 | 1.43 |
| France | 0.48 | 1.87 |
| Germany | 1.41 | 3.81 |
| Greece | 0.02 | 0.09 |
| Hungary | 0.40 | 0.92 |
| Ireland | 0.14 | 0.28 |
| Italy | 0.53 | 2.08 |
| Latvia | 0.23 | 1.22 |
| Lithuania | 0.21 | 0.66 |
| Luxembourg | 1.55 | 4.03 |
| Malta | 0.70 | 2.73 |
| Netherlands | 1.15 | 3.14 |
| Poland | 0.53 | 1.44 |
| Portugal | 0.06 | 0.35 |
| Romania | 0.10 | 0.45 |
| Slovakia | 0.79 | 1.80 |
| Slovenia | 1.42 | 3.37 |
| Spain | 0.06 | 0.28 |
| Sweden | 1.10 | 0.65 |
| United Kingdom | 0.48 | 0.12 |

The authors recognised problems with the use of their results, for example:

- The approach assumes that the cost of replacing an ecosystem or its services is an estimate of the value of the ecosystem or its services. To the extent that restoration is applied, the cost of restoration can only be regarded as a minimum estimate. To the extent that restoration is not applied, a conclusion may be that the costs exceed the benefits of action, that benefits have been underestimated (qualitatively or quantitatively), or simply that funding for restoration is unavailable.

- The approach is not based on individual preferences but on an ecological or expert standard and the cost to re-establish this standard.

- Proposed interventions may not be a perfect substitute for the lost ecosystem service, e.g. existence values of certain species or ecosystems are not replaceable.

- Some damage may not be recognised immediately, or, like the effects of gradual eutrophication of ecosystems, may take many years to reach steady state.

- Restoration may bring a variety of benefits that are not recognised through the restoration costs (Pearce and Moran, 1994).

The use of damage cost data implies that the marginal response to a change in deposition will be to alter the level of restoration being undertaken. For the analysis presented below, this assumption seems reasonable at least for the protection of Natura 2000 sites which should be ensured as a consequence of existing legislation.

The question remains as to how much of the impact on 'biodiversity' is valued using the repair cost approach. For estimation of externalities of air pollution on buildings, including buildings of cultural merit, Rabl (1999) and Rabl et al (2014) argue that the social optimum is to undertake action when the repair costs are equivalent to loss of amenity. The total cost is then twice the repair cost. This could be factored into the current analysis by presenting a range, with the upper bound double the estimate based on the approach used by Ott et al. Of course, this assumption implies that we have some understanding of amenity losses, at least in qualitative terms.

The European Nitrogen Assessment (Sutton et al, 2011) provides a thorough review of the sources, flows and impacts of nitrogen in the European environment. Chapter 22 of the Assessment (Brink et al, 2011) provides information on costs and benefits. Results are shown in Table 5. The lower bound for ecosystem damage (also adopted as the best estimate) was based on the costs of restoring biodiversity loss due to reactive nitrogen deposition estimated by Ott et al (2006). The upper bound was arbitrarily set 5 times higher as a possible value when using an ecosystem service approach.

**Table 5. Unit damage costs as €/kg reactive nitrogen for the major $N_r$ pollutants. Upper figures, best estimate, lower figures range. From Brink et al (2011).**

| Pollutant | Health | Ecosystems | Climate | Total |
|---|---|---|---|---|
| $N_r$ to water | 1<br>0 – 4 | 12<br>5 – 20 | | 13<br>5 – 24 |
| $NH_3$-N to air | 12<br>2 – 20 | 2<br>2 – 10 | | 14<br>4 – 30 |
| NOx-N to air | 18<br>10 – 30 | 2<br>2 – 10 | | 20<br>12 – 40 |
| $N_2O$-N to air | 2<br>1 – 3 | | 9<br>5 – 15 | 11<br>6 – 18 |

There is a question whether the results of Ott et al are consistent with this table, as they are expressed per kg of NOx, $NH_3$ deposited, rather than emitted. This is moderated in the table by the selection of the lower end of the range as the best estimate.

These estimates for ecosystem damage, being based around the Ott et al study, suffer the same deficiencies. Further to this, the figures adopted do not differentiate by location of source within Europe, for which there can be a great deal of variation. If, for example, nitrogen is released in an area with little or no exceedance of critical loads, damage is anticipated to be zero or close to it. For health impacts variation of about a factor 10 has been identified between average damage costs for different countries for various pollutants (EEA, 2011) and the same may well be true for impacts to ecosystems.

As an aside, it is noted that the health values given by Brink et al were taken from ExternE (2005). These values would ideally need revision now, following work undertaken by WHO-Europe on the REVIHAAP and HRAPIE studies (WHO, 2013 a, b) and changes made to the EMEP modelling. The costs of climate change were based on a range of €10 – 30 tonne $CO_2$, reflecting variation in the carbon price since 2005. Again, alternative positions are available, for example using results from Stern (2006). The European Environment Agency has recently adopted a range of €9.5 to 38.1/t $CO_2$ (2005 prices) reflecting values for the EU Emissions Trading Scheme used in policy modelling by the European Commission (EEA, 2014), values that are not dissimilar to those used by Brink et al.

# 3 Overview of methods for analysis

## 3.1 Broad approaches for quantification

Following a workshop of this Work Package held at RIVM in December 2013, it was decided to investigate the use of three different approaches to the quantification of damage to natural ecosystems.

The first of these is based on available information from stated preference studies on 'willingness to pay' (WTP) for protection of biodiversity, drawing on Christie et al (2011) and Jones et al (2013). The second is based on use of repair costs drawing on the results of Ott et al (2006), and the third on the inferred costs of environmental policies. The methods based on repair costs and inferred costs of policies are both 'revealed preference' approaches.

It was also decided that the work should focus initially on Natura 2000 areas, for which there is a legal responsibility on Member States to preserve, maintain and restore.

Each of the methods is applied in the following three chapters, with results considered together at the end of the report.

It is stressed at this point that the stated preference WTP methods forming the first approach are the preferred option for quantification as they best reflect accepted environmental economic methodology (including those used for quantification of health benefits by Holland, 2014). However, data availability is very limited, and so it is useful also to test the other approaches to see what additional information they can provide.

## 3.2 WTP for protection of biodiversity (stated preference)

The analysis here is based on the results of the Christie et al study (2006, 2010, 2011) designed to inform the development and appraisal of the UK's biodiversity Action Plan (UK BAP). Christie's work provides estimates of household WTP for environmental protection, and hence reflect preference from the perspective of the general public.

Of the studies available it is considered most appropriate to the needs of the present work for several reasons, for example:

1. it deals with WTP for a change in status of the ecosystems under investigation
2. It recognises that different types of ecosystem will be valued differently
3. It is aligned to a degree with the ecosystem services concept
4. It is European.

Against this, it is a single study (albeit part of a series), investigating valuation in a single country. Accepting this, there is a need to consider whether there is information available that would improve the process of value transfer from the UK to the whole EU. Different types of data are considered, below, for this process, including information on GDP/capita and environmental expenditures.

### 3.3 Restoration costs (revealed preference)

Ecological restoration is applied extensively already, for example to the Dutch Heathlands. The concept behind this approach is that the costs of restoration reflect societal WTP for improved protection of biodiversity.

To the extent that restoration is undertaken it can reasonably be said that associated costs provide some insight, but only as a minimum estimate (minimum on the basis that WTP has to be at least as large as the restoration cost for action to be carried out, but could be larger). The need to restore indicates that some level of damage has occurred that society finds unacceptable, and this has a cost: one would not restore for the simple sake of restoration. Rabl (1999) considers a similar issue for buildings maintenance, considering the amenity loss from soiling. If restoration costs were higher than the amenity loss it is argued that the restoration would not be performed. Total costs (restoration + amenity) can, however, be minimised by restoring at the point where amenity losses are equal to restoration costs. Hence, assuming society acts to minimise cost:

Total damage cost = 2 x restoration cost

Amenity loss by time of restoration = restoration cost

Of course, soiling of buildings and eutrophication damage to ecosystems have a number of differentiating characteristics. Not least is the fact that building soiling will be witnessed day in, day out until the building occupier decides that action is needed. For ecosystems, however, damage may go undetected for a long time. This may imply that Rabl's factor 2 is likely to be too low. This adds some weight to the factor 5 adopted by Brink and van Grinsven.

There are several problems with the restoration cost approach, including:

- The concept of restoration is questionable. After restoration an ecosystem may look the same as it was originally. However, even where species are reintroduced successfully there will be a loss of genetic stock. This can bring its own problems: a restricted genetic stock has been identified as a major risk factor for the spread of certain diseases including Ash Dieback. The fine detail (for example the range of soil invertebrates present) will also be different. Hence there is a real question of the extent to which restoration goes beyond the cosmetic.

- The definition of 'restored' requires some reference position to be adopted. This may require some rather arbitrary decisions to be taken (as implied by Ott et al's referencing to the Swiss lowlands). One possible reference point might be thought of as the date at which legislation such as the Birds and

Habitats Directives were passed. However, the driver for this legislation was environmental degradation, so conditions when the Directives were passed will not represent the undamaged state.

- There is a question of whether society will think that restoration is worthwhile. If a site is not restored the WTP for its protection can be concluded to be lower than the restoration cost. This problem is partially overcome by limiting analysis to Natura 2000 sites because of the legal mandate that they should not be allowed to deteriorate, which is not met whilst critical loads and levels are exceeded.

However, seen as providing an alternative perspective to that gained from the other methods used here, this approach is accepted for the purposes of the present report.

## 3.4 Regulatory revealed preference

This third approach takes the view that the costs of meeting critical loads in Natura 2000 areas are implied in the Birds and Habitats Directives. Continued emissions at current levels will lead to continued exceedance, and hence the requirement that Natura 2000 sites should not be allowed to deteriorate will not be met. Alongside the more local $NH_3$ deposition there is a need to also consider (large scale) NOx-reduction.

The valuation can be carried out by considering the costs of scenarios modelled using GAINS and the extent to which N emissions need to be reduced to meet the critical load for eutrophication. In some areas it is possible that rather small reductions in emissions may be sufficient, but in many the required emission reduction may exceed what is possible using IIASA's Maximum Technically Feasible Reduction scenario. The potential for smaller reductions in emissions in some is could be investigated in subsequent scenario analysis with the GAINS model. Some of the measures included in the GAINS model are extremely expensive per unit emission avoided. A further sensitivity that could be explored would be the possibility that a small level of exceedance (say 10% or 20%) could be easily avoided through nature management measures.

Of course, our analysis deals with the costs of environmental damage rather than the costs of abatement in each country. This should ideally be recognised in the way that inferred damage costs are attributed to each country.

As will be shown, application of all abatement measures within GAINS is insufficient to attain full compliance with critical loads. Then, the costs of additional measures need to be considered. A possibility would be to control livestock farming in affected areas, and some estimate of the cost of this can be made from data on the annual value of livestock production.

Again, there are several potential problems with this approach, for example:

- The Birds Directive was first adopted in 1979, and the Habitats Directive in 1992. It is not clear to what extent the policy makers involved in this process were aware of the threats posed by nitrogenous air pollutants.

- It is unclear to what extent subsequent revisions of the Directives have paid attention to damage caused by air pollution, though the 7th Environmental Action Programme of the European Commission reconfirms the need of meeting critical loads (no significant damage to ecosystems) by 2050. However, to the extent that some policy makers may have been aware of the threat of air pollution, they may have regarded it as being dealt with directly through air pollution legislation and hence not of their concern.

- Existing air quality polices are developed against health, as well as ecological objectives.

Again, issues are partially offset by considering the approach not in isolation, but as providing an alternative perspective to the methods discussed above. The approach is also useful for stressing the difficulty in ensuring the health of sites designated for protection given the burdens imposed by transboundary air pollution.

# 4 Willingness to pay approach

## 4.1 Methods and data

As noted above, this approach is based on use of WTP estimates for ecological protection taken from the work of Christie et al (2011) which provides the valuation input to the work of Jones et al. It should be noted that at the present time it has not proved possible to factor species richness (Figure 6) into the analysis, though this work continues. Although the valuation work is only from the UK it is considered here to be the most relevant analysis based on accepted environmental economic methods that is available from the European literature, given the fact that it considers WTP for specific changes in ecological protection and considers a variety of types of ecosystem.

The selection of the appropriate WTP estimate from Christie et al is open to debate. For the purposes of illustration, this paper takes the estimate of €10-30 per household per year adopted for analysis by Maas (2014) for presentation to the 2014 meeting of ICP Mapping and Modelling. This is representative of WTP for 'non-charismatic' species in the area local to one's home ('within own region' as expressed by Christie et al), selected also by Jones et al as an indicator of the WTP to protect 'biodiversity'.

There are two ways that this figure could be extrapolated to other countries (leaving aside, for the moment, the major issues of value transfer here relating to different incomes in different countries and differences in appreciation of nature between countries). The first is to assume that an average household is willing to pay €10-30 per household per year for nature protection, the second that WTP will vary according to the area of ecosystems at risk, in line with the equivalent UK valuation per unit area. Applying results per household suggested very high valuations per unit area in some countries where there was limited exceedance of the critical load over nature areas. Accordingly, a valuation of €80 to 240/ha/yr was calculated, applying the household WTP described above to the area of protected UK sites at risk. This was applied to protected sites at risk in all countries, using data from the Current Legislation (CLE) and Maximum Technically Feasible Reduction (MTFR) scenarios for 2025 and 2030 considered in the development of the Clean Air Policy Package of December 2013. No consideration was given to unprotected sites, recognising that the Christie et al work was performed against the background of the UK's Biodiversity Action Plan. Further refinements to the analysis are discussed below.

## 4.2 Results

Results based on applying a common value per hectare of protected land at risk through exceedance of the critical load for eutrophication of €80 are shown in Table 6. Table 7 is similar, but takes a valuation of €240/ha reflecting the factor 3 variation across the range cited above.

**Table 6. Estimated costs of exceedance of critical loads over Natura 2000 sites in Europe for estimated WTP of €80/ha. Italicised lines show countries for which data on protected areas are approximated.**

| Country | CLE, €M/year | MTFR, €M/year | Benefit, €M/year | | CLE, €M/year | MTFR, €M/year | Benefit, €M/year |
|---|---|---|---|---|---|---|---|
| | 2025 | 2025 | 2025 | | 2030 | 2030 | 2030 |
| *Austria* | *46* | *17* | *30* | | *43* | *14* | *29* |
| *Belgium* | *0* | *0* | *0* | | *0* | *0* | *0* |
| *Bulgaria* | *101* | *82* | *19* | | *101* | *82* | *19* |
| Croatia | 0 | 0 | 0 | | 0 | 0 | 0 |
| Cyprus | 6 | 6 | 0 | | 6 | 6 | - |
| Czech Rep. | 6 | 3 | 2 | | 5 | 3 | 3 |
| Denmark | 13 | 12 | 1 | | 13 | 12 | 1 |
| Estonia | 13 | 7 | 6 | | 13 | 7 | 7 |
| Finland | 5 | 4 | 2 | | 5 | 4 | 1 |
| France | 700 | 401 | 299 | | 678 | 379 | 299 |
| Germany | 329 | 194 | 135 | | 321 | 183 | 137 |
| Greece | 132 | 128 | 3 | | 131 | 128 | 3 |
| Hungary | 84 | 71 | 13 | | 82 | 71 | 11 |
| Ireland | 0 | 0 | 0 | | 0 | 0 | 0 |
| Italy | 252 | 150 | 102 | | 242 | 139 | 103 |
| Latvia | 34 | 27 | 8 | | 34 | 26 | 8 |
| Lithuania | 43 | 39 | 4 | | 43 | 39 | 4 |
| Luxembourg | 3 | 2 | 0 | | 3 | 2 | 0 |
| Malta | 0 | 0 | 0 | | - | - | - |
| Netherlands | 31 | 28 | 3 | | 31 | 28 | 4 |
| *Poland* | *305* | *206* | *99* | | *301* | *193* | *108* |
| Portugal | 74 | 66 | 8 | | 74 | 66 | 8 |
| Romania | 163 | 146 | 17 | | 162 | 143 | 19 |
| Slovakia | 75 | 67 | 8 | | 74 | 65 | 9 |
| Slovenia | 11 | 2 | 9 | | 9 | 1 | 8 |
| Spain | 703 | 617 | 86 | | 701 | 605 | 96 |
| Sweden | 9 | 6 | 3 | | 8 | 5 | 3 |
| *United Kingdom* | *36* | *12* | *24* | | *35* | *11* | *24* |
| **EU-28** | **3175** | **2294** | **881** | | **3,116** | **2,211** | **905** |

Some consideration is given below as to how these figures may be adjusted for different countries. However, it is noted already that further thought could be given as to whether this should just apply to Natura 2000 areas only, or to all ecosystems, and whether it is possible to define a price elasticity according to abundance per capita (with increased abundance generating a lower value per hectare). In favour of adoption of a standard price across the EU is the fact that Natura 2000 is essentially transboundary in defining 'European heritage'. Against this view, there is very limited (via the LIFE programme) funding for management of Natura 2000 at the EU level. Further consideration will be given to price elasticity, though it requires further data than were available here.

**Table 7. Estimated costs of exceedance of critical loads over Natura 2000 sites in Europe for estimated WTP of €240/ha. Italicised lines show countries for which data on protected areas are approximated.**

| Country | CLE, €M/year | MTFR, €M/year | Benefit, €M/year | | CLE, €M/year | MTFR, €M/year | Benefit, €M/year |
|---|---|---|---|---|---|---|---|
| | 2025 | 2025 | 2025 | | 2030 | 2030 | 2030 |
| *Austria* | *239* | *131* | *108* | | *230* | *123* | *106* |
| *Belgium* | *1* | *0* | *0* | | *1* | *0* | *0* |
| *Bulgaria* | *308* | *251* | *57* | | *308* | *251* | *57* |
| Croatia | 6 | 4 | 3 | | 6 | 3 | 3 |
| Cyprus | 32 | 31 | 1 | | 32 | 31 | 1 |
| Czech Rep. | 30 | 16 | 14 | | 29 | 15 | 15 |
| Denmark | 44 | 41 | 3 | | 43 | 40 | 3 |
| Estonia | 741 | 423 | 318 | | 718 | 399 | 320 |
| Finland | 345 | 206 | 140 | | 336 | 194 | 142 |
| France | 2232 | 1331 | 902 | | 2166 | 1265 | 901 |
| Germany | 1072 | 654 | 418 | | 1043 | 620 | 423 |
| Greece | 395 | 386 | 10 | | 394 | 384 | 10 |
| Hungary | 505 | 362 | 142 | | 487 | 352 | 136 |
| Ireland | 35 | 27 | 8 | | 35 | 27 | 8 |
| Italy | 799 | 489 | 310 | | 770 | 457 | 313 |
| Latvia | 105 | 82 | 24 | | 104 | 81 | 23 |
| Lithuania | 129 | 117 | 12 | | 129 | 117 | 12 |
| Luxembourg | 39 | 35 | 4 | | 39 | 34 | 4 |
| Malta | 305 | 206 | 99 | | 301 | 193 | 108 |
| Netherlands | 167 | 151 | 17 | | 167 | 148 | 19 |
| *Poland* | *1079* | *764* | *315* | | *1065* | *721* | *343* |
| Portugal | 296 | 266 | 30 | | 295 | 262 | 33 |
| Romania | 499 | 440 | 59 | | 494 | 430 | 64 |
| Slovakia | 927 | 818 | 109 | | 921 | 799 | 123 |
| Slovenia | 40 | 11 | 29 | | 36 | 9 | 27 |
| Spain | 2146 | 1863 | 284 | | 2137 | 1825 | 313 |
| Sweden | 3201 | 2311 | 890 | | 3141 | 2227 | 914 |
| *United Kingdom* | *109* | *36* | *73* | | *106* | *33* | *73* |
| **EU-28** | **9525** | **6882** | **2643** | | **9347** | **6633** | **2714** |

## 4.3 Adjustment of values by Member State

### 4.3.1 The need for differentiation by Member State

The need to differentiate unit values by Member State is dependent on the scope of the analysis. For analysis at the European Union level it is common practice to apply uniform values across the EU (as in the analysis of the benefits of reducing air pollution by Holland, 2014, for the Clean Air Policy Package). The assumption of different valuations in different places would run counter to the ‘level playing field’ philosophy that underpins much EU environmental regulation. On this basis, for decisions made at the EU level, the valuation of health should be consistent throughout the Union, as should valuation of concern for the environment.

However, the same does not apply to valuation for policies implemented at the national level. In that situation it is appropriate to use valuations that reflect national, rather than continental preference. Consider first the situation in a country that has a high WTP for environmental protection. There is no basis for dictating to that country that they should limit environmental expenditure to a level representative of the Union as a whole. Consider then a country that has a low WTP for environmental protection because of low average income. Why should the priorities for protection adopted elsewhere be assumed applicable here also, given that rich and poor may have very different ideas as to what is the most effective use of available resource?

It may be asked whether the proposed use of different approaches at different scales is inconsistent. It is here considered that this is not the case. If the EU as a whole considers that additional environmental protection is desirable it may provide funding for such measures, for example through the EU's LIFE Programme, funding for which is equal to €398 million for 2014[9].

The factors considered here to provide a basis for adjustment are:

- Average per capita income, reflected through GDP adjusted for purchasing power parity,
- Levels of environmental concern as indicated by Eurobarometer survey data and
- Government expenditure on environmental protection, also as an indication of societal concern.

### 4.3.2 Adjustment by average per capita income

Table 8 shows variation in GDP per capita across Europe, expressed in purchasing power standards (PPS). PPS adjustment factors out differences in buying power per unit of currency between countries.

The standard approach for adjustment for income between countries to derive an estimate of the value of a good at a policy site (Vp) is to multiply the unit valuation from the original study site (Vs) by the ratio of income in the new country (Ip) divided by income for the original country (Is), raised to the power of the income elasticity of demand for the environmental good in question (ß):

$$V_p = V_s (I_p/I_s)^{ß}$$

The income elasticity of demand shows how willingness to pay varies with income. For mortality, OECD (2012a) recommends using an elasticity of 0.8, with a sensitivity analysis using 0.4. The use of an elasticity less than 1 means that people on lower incomes are, as an average across society, willing to pay a greater share of their income on health protection than richer people. However, there is evidence that people see the environment as a luxury good (see, for

[9] http://ec.europa.eu/environment/funding/pdf/awp_2014.pdf.

example, Martinez-Alier, 1995) where the opposite would apply, that WTP as a fraction of one's income would increase with income. This seems appropriate in relation to pollution impacts where individuals have very little control over impacts (the opposite position to interventions to protect one's own health) and lifestyles are distanced to some extent from ecological quality. Given conflicting evidence on the magnitude of the elasticity, a core value of 1 and sensitivity value of 1.5 are therefore adopted in the analysis shown at the end of this section.

**Table 8. Variation in GDP per capita, purchasing power standards (EU28 = 100), for 2005, 2010 and 2012**

| | 2005 | 2010 | 2012 |
|---|---|---|---|
| **EU (28 countries)** | **100** | **100** | **100** |
| | | | |
| Austria | 125 | 127 | 130 |
| Belgium | 120 | 121 | 120 |
| Bulgaria | 37 | 44 | 47 |
| Croatia | 57 | 59 | 62 |
| Cyprus | 93 | 97 | 92 |
| Czech Republic | 79 | 81 | 81 |
| Denmark | 124 | 128 | 126 |
| Estonia | 62 | 64 | 71 |
| Finland | 114 | 114 | 115 |
| France | 110 | 109 | 109 |
| Germany | 116 | 120 | 123 |
| Greece | 91 | 88 | 75 |
| Hungary | 63 | 66 | 67 |
| Ireland | 144 | 129 | 129 |
| Italy | 105 | 103 | 101 |
| Latvia | 50 | 55 | 64 |
| Lithuania | 55 | 62 | 72 |
| Luxembourg | 254 | 263 | 263 |
| Malta | 80 | 87 | 86 |
| Netherlands | 131 | 130 | 128 |
| Poland | 51 | 63 | 67 |
| Portugal | 80 | 80 | 76 |
| Romania | 35 | 48 | 50 |
| Slovakia | 60 | 74 | 76 |
| Slovenia | 87 | 84 | 84 |
| Spain | 102 | 99 | 96 |
| Sweden | 122 | 124 | 126 |
| United Kingdom | 124 | 108 | 106 |
| | | | |
| **Non EU** | | | |
| Albania | 22 | 26 | 30 |
| Bosnia and Herzegovina | 25 | 29 | 29 |
| Iceland | 130 | 115 | 115 |
| Montenegro | 31 | 42 | 41 |
| Norway | 178 | 181 | 195 |
| Serbia | 32 | 35 | 36 |
| Switzerland | 137 | 152 | 158 |
| TFYR Macedonia | 29 | 36 | 35 |

| | | | |
|---|---|---|---|
| Turkey | 42 | 50 | 54 |

A further factor to take into account given the timescales that are relevant to ECLAIRE, is differential change in income across the EU between now and the middle of the century. OECD (2012b) provides some insight on trends (see Figure 7) with disparity in GDP per capita forecast to close significantly for many of the lower income EU Member States. Global GDP is forecast to grow at 3% per annum, with a lower rate of 2% in OECD countries.

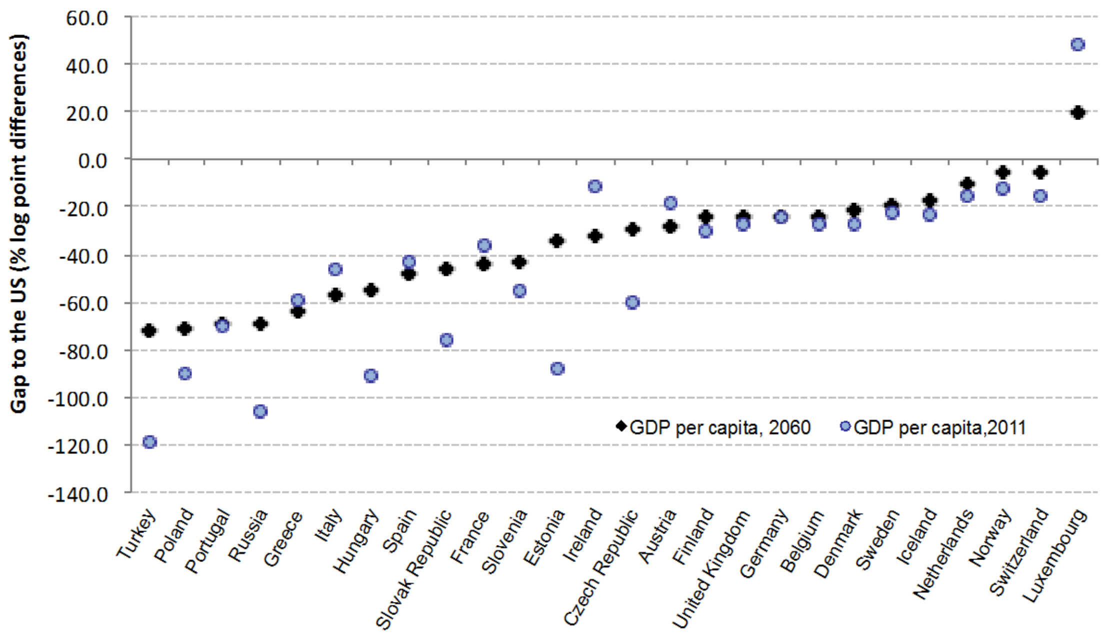



**Figure 7. Change in GDP per capita relative to the USA. Data source: OECD, 2012b.**

This reduction in economic disparity across the EU will lead to an increase in valuations by the time of the ECLAIRE scenarios. This should be accounted for in the analysis, though requires conclusions to be reached on the appropriate level of income elasticity. Such adjustment is not carried out below but should be considered for the future.

Results are presented, adjusted for income, in Section 4.3.5.

### 4.3.3 Levels of environmental concern

Eurobarometer surveys provide an indication of public attitude across the EU. Special Eurobarometer 365 was focused on attitudes to the environment. Results indicate that more than 90% of people in all countries considered environment to be very or fairly important. There is more differentiation in the results when considering the 'very' and 'fairly' important categories separately, with results for 'very important' ranging from 44% (Austria and Finland) to 89% (Cyprus). Ignoring the strength of response it is notable that 95% of respondents consider the environment to be important.

**Table 9. National results of survey: How important is the environment to you personally? (adapted from Eurobarometer, 2011)**

| Importance | Very | Fairly | Not very | Not at all | Don't know | Total important | Total not important |
|---|---|---|---|---|---|---|---|
| Austria | 44 | 46 | 8 | 2 | 0 | 90 | 10 |
| Belgium | 62 | 33 | 4 | 1 | 0 | 95 | 5 |
| Bulgaria | 79 | 19 | 1 | 0 | 1 | 98 | 1 |
| Cyprus | 89 | 11 | 0 | 0 | 0 | 100 | 0 |
| Czech Republic | 60 | 35 | 5 | 0 | 0 | 95 | 5 |
| Denmark | 60 | 36 | 4 | 0 | 0 | 96 | 4 |
| Estonia | 54 | 40 | 5 | 0 | 1 | 94 | 5 |
| Finland | 44 | 49 | 6 | 1 | 0 | 93 | 7 |
| France | 62 | 35 | 2 | 1 | 0 | 97 | 3 |
| Germany | 51 | 44 | 4 | 1 | 0 | 95 | 5 |
| Greece | 70 | 28 | 2 | 0 | 0 | 98 | 2 |
| Hungary | 64 | 32 | 4 | 0 | 0 | 96 | 4 |
| Ireland | 60 | 34 | 4 | 2 | 0 | 94 | 6 |
| Italy | 61 | 33 | 4 | 2 | 0 | 94 | 6 |
| Latvia | 53 | 42 | 4 | 1 | 0 | 95 | 5 |
| Lithuania | 59 | 35 | 5 | 0 | 1 | 94 | 5 |
| Luxembourg | 72 | 24 | 3 | 0 | 1 | 96 | 3 |
| Malta | 86 | 14 | 0 | 0 | 0 | 100 | 0 |
| Netherlands | 50 | 43 | 6 | 1 | 0 | 93 | 7 |
| Poland | 47 | 45 | 5 | 1 | 2 | 92 | 6 |
| Portugal | 51 | 44 | 5 | 0 | 0 | 95 | 5 |
| Romania | 56 | 36 | 5 | 1 | 2 | 92 | 6 |
| Slovakia | 57 | 38 | 4 | 0 | 1 | 95 | 4 |
| Slovenia | 80 | 18 | 1 | 1 | 0 | 98 | 2 |
| Spain | 56 | 38 | 4 | 1 | 1 | 94 | 5 |
| Sweden | 83 | 15 | 1 | 1 | 0 | 98 | 2 |
| UK | 58 | 36 | 5 | 1 | 0 | 94 | 6 |
| **EU 27** | **58** | **37** | **4** | **1** | **0** | **95** | **5** |

Adjustment of values by the expressed levels of environmental concern raises the question of what factors cause people to consider the environment as very important. There seems to be potential for rather low numbers expressing a high level of importance in two very contrasting situations:

- Countries where environmental quality is good
- Countries where environmental quality is poor, but policy debates are focused on other issues

In the first case it is quite possible that environmental quality is good because citizens have a high WTP for environmental protection. In the second, there should be a lower WTP. Hence adjustment according to the results of Table 9 may generate some misleading results and is not adopted.

Eurobarometer also asked people to state what they thought of when people spoke of 'the environment'. Results are shown in Table 10. Respondents were split into two groups who were asked the same questions with one exception. Group A were asked to consider 'Protecting nature' and Group B were asked to consider 'Protecting biodiversity'. Results are starkly different, with Group A ranking 'Protecting nature' most highly and Group B ranking 'Protecting biodiversity' lowest. This outcome is relevant quite generally to the work of ECLAIRE, in terms of needing to communicate the nature of impacts effectively with members of the public. It also raises issues for the effective construction and framing of surveys to investigate willingness to pay for environmental protection.

**Table 10. Results of survey: What do you think of when people refer to 'the environment'? (adapted from Eurobarometer, 2011)**

| | Group A | Group B |
|---|---|---|
| Protecting nature (Group A)/biodiversity (Group B) | 47% | 20% |
| The state of the environment our children will inherit | 41% | 45% |
| Climate change | 40% | 41% |
| Pollution in towns and cities | 39% | 42% |
| Man made disasters (oil spills, etc.) | 39% | 41% |
| The quality of life where you live | 33% | 37% |
| Using up natural resources | 31% | 34% |
| Green and pleasant landscapes | 28% | 31% |
| Natural disasters (earthquakes, floods, etc.) | 26% | 28% |
| Other / don't know | 4% | 4% |

### 4.3.4 Government expenditure on environmental protection

Eurostat (2014) provides information on environmental expenditures by business and government. The former are likely to have expenditure dominated by legislated requirements (e.g. emission controls to meet regulatory limits) and so have not been considered. Data for expenditures by government are included below, however (Table 11). Results show a great deal of variation, reasons for which are unclear. For example, the country with the highest government expenditure per capita is Luxembourg according to this list, with average expenditure more than 5 times higher than Germany. This might suggest inconsistency in the way that data are reported. For this reason it seems inappropriate to apply the data to adjust the earlier WTP estimates.

In the near future (2017) systematic information will be generated by EU Member States through the Environmental Protection Expenditure Accounts (EPEA) required under EU Regulation 691/2011[10]. Some countries have already generated estimates using a similar format, an example for Austria being shown in Table 12. This might provide a basis for adjustment in the future, but at the present time too few countries have provided data for its application here.

[10] http://eur-lex.europa.eu/legal-content/EN/TXT/?uri=CELEX:52013PC0247

## Table 11. Environmental protection expenditure in Europe by government, Euro per capita. Source: Eurostat, 2014

| | 2011 | Average 2002-2004 | Average 2005-2007 | Average 2008-2010 | Average 2002-2011 |
|---|---|---|---|---|---|
| **EU27** | **166** | **131** | **151** | **169** | **152** |
| | | | | | |
| Austria | | 210 | 237 | 198 | 217 |
| Belgium | | 160 | 150 | 150 | 153 |
| Bulgaria | 31 | 8 | 15 | 27 | 18 |
| Croatia | 32 | 8 | 16 | 4 | 12 |
| Cyprus | | 57 | | | 57 |
| Czech Republic | 76 | | 52 | 61 | 61 |
| Denmark | | 251 | 251 | 283 | 255 |
| Estonia | | 14 | 18 | 22 | 18 |
| Finland | | 164 | 175 | 201 | 180 |
| France | | 143 | 162 | 207 | 170 |
| Germany | | 105 | 99 | 99 | 101 |
| Greece | | | | | |
| Hungary | 39 | 50 | 54 | 33 | 45 |
| Ireland | | | | | |
| Italy | 229 | 202 | 208 | 224 | 213 |
| Latvia | | 5 | 60 | 74 | 46 |
| Lithuania | | 11 | 53 | 96 | 59 |
| Luxembourg | 655 | 532 | 570 | 569 | 567 |
| Malta | 230 | 137 | 197 | 262 | 202 |
| Netherlands | | 452 | 509 | 516 | 497 |
| Poland | 51 | 17 | 30 | 41 | 32 |
| Portugal | 78 | 68 | 73 | 89 | 77 |
| Romania | 59 | 5 | 22 | 39 | 26 |
| Slovakia | 40 | 11 | 22 | 32 | 23 |
| Slovenia | | 134 | 117 | 157 | 136 |
| Spain | | 51 | 68 | 72 | 64 |
| Sweden | 139 | 101 | 134 | 123 | 121 |
| UK | | 139 | | | 139 |
| | | | | | |
| **Non-EU** | | | | | |
| Iceland | | 118 | | | 118 |
| Norway | 485 | 261 | 334 | 406 | 349 |
| Switzerland | | 334 | | | 334 |
| Turkey | | 12 | 25 | 31 | 23 |

### Table 12. EPEA account for Austria.

Financing of domestic environmental protection expenditure 2011 in million €

| Financing Units | Protection of ambient air and climate | Wastewater management | Waste management | Protection and remediation of soil, groundwater and surface water | Noise and vibration abatement | Protection of biodiversity and landscape | Protection against radiation | R&D Research and development | Other environmental protection activities |
|---|---|---|---|---|---|---|---|---|---|
| Public sector | 101.4 | 350.2 | 60.4 | 10.7 | 1.1 | 67.7 | 0.1 | 0.6 | 21.6 |
| Non governmental organisations | 14.6 | 1.6 | 4.0 | 9.4 | 0.7 | 131.4 | 0.0 | 3.4 | 26.2 |
| Households | 187.4 | 679.1 | 1 123.6 | 0.0 | 142.4 | 17.6 | 0.0 | 0.0 | 602.7 |
| Enterprises | 504.9 | 1 443.9 | 2 700.9 | 1 279.1 | 68.1 | 543.8 | 0.0 | 187.9 | 931.7 |
| **Total** | **808.3** | **2 474.7** | **3 888.9** | **1 299.2** | **212.3** | **760.5** | **0.1** | **192.0** | **1 582.1** |

S: STATISTICS AUSTRIA, Enviornmental protection expenditure accounts, on behalf of BMLFUW. Compiled on 14 December 2013.

Source:
http://www.statistik.at/web_en/statistics/energy_environment/environment/environmental_protection_expenditure_accounts_epea/index.html

### 4.3.5 Appling adjustment by income

Given the lack of suitable data to adjust the WTP based valuations of Table 6 and Table 7 by some differential of environmental concern between countries, the only form of adjustment applied here is for differences in income. To illustrate the level of variation that this induces we need use only one set of results (Table 6, based on WTP of €80/ha) as the variability identified for that case will be the same proportionally as for the case using a higher baseline estimate of WTP (€240/ha). Results in Table 13 demonstrate the effects of adjustment using the variation in GDP/capita from Table 8 assuming an income elasticity of 1 and 1.5. Adjusted values are lower than the original estimates, but by less than 10%.

### Table 13. Damage and benefit estimates for the EU28 calculated using a WTP of €80/ha for the UK, and income adjusted values with income elasticity of 1 and 1.5.

| | CLE, €M/year | MTFR, €M/year | Benefit, €M/year | | CLE, €M/year | MTFR, €M/year | Benefit, €M/year |
|---|---|---|---|---|---|---|---|
| **EU28** | **2025** | **2025** | **2025** | | **2030** | **2030** | **2030** |
| Unadjusted | 3,175 | 2,294 | 881 | | 3,116 | 2,211 | 905 |
| Adjusted, elasticity = 1 | 2,793 | 1,961 | 832 | | 2,735 | 1,884 | 851 |
| Adjusted, elasticity = 1.5 | 2,678 | 1,856 | 822 | | 2,621 | 1,781 | 840 |

Whilst the aggregated EU28 results show little difference (<10%) as a result of adjustment, there is much more variability between Member States (Table 14). There is also significant variation introduced through different assumptions on elasticity, with variation for some countries in excess of +/-30% (Table 15).

**Table 14. Comparison of results for 2025. Left hand side, unadjusted for income, right hand side, adjusted for income with income elasticity of 1.5.**

| Country | CLE, €M/year | MTFR, €M/year | Benefit, €M/year | | CLE, €M/year | MTFR, €M/year | Benefit, €M/year |
|---|---|---|---|---|---|---|---|
| | 2025 | 2025 | 2025 | | 2025 | 2025 | 2025 |
| | | Unadjusted | | | | Adjusted | |
| Austria | 46 | 17 | 30 | | 63 | 22 | 40 |
| Belgium | 0 | 0 | 0 | | 0 | 0 | 0 |
| Bulgaria | 101 | 82 | 19 | | 30 | 24 | 6 |
| Croatia | 0 | 0 | 0 | | 0 | 0 | 0 |
| Cyprus | 6 | 6 | 0 | | 5 | 5 | 0 |
| Czech Rep. | 6 | 3 | 2 | | 4 | 2 | 2 |
| Denmark | 13 | 12 | 1 | | 17 | 16 | 1 |
| Estonia | 13 | 7 | 6 | | 7 | 4 | 3 |
| Finland | 5 | 4 | 2 | | 6 | 4 | 2 |
| France | 700 | 401 | 299 | | 730 | 418 | 312 |
| Germany | 329 | 194 | 135 | | 412 | 243 | 168 |
| Greece | 132 | 128 | 3 | | 78 | 76 | 2 |
| Hungary | 84 | 71 | 13 | | 42 | 36 | 7 |
| Ireland | 0 | 0 | 0 | | 0 | 0 | 0 |
| Italy | 252 | 150 | 102 | | 234 | 140 | 95 |
| Latvia | 34 | 27 | 8 | | 16 | 12 | 4 |
| Lithuania | 43 | 39 | 4 | | 24 | 22 | 2 |
| Luxembourg | 3 | 2 | 0 | | 10 | 9 | 1 |
| Malta | 0 | 0 | 0 | | 0 | 0 | 0 |
| Netherlands | 31 | 28 | 3 | | 41 | 37 | 4 |
| Poland | 305 | 206 | 99 | | 153 | 104 | 50 |
| Portugal | 74 | 66 | 8 | | 45 | 40 | 5 |
| Romania | 163 | 146 | 17 | | 53 | 47 | 5 |
| Slovakia | 75 | 67 | 8 | | 45 | 41 | 5 |
| Slovenia | 11 | 2 | 9 | | 7 | 1 | 6 |
| Spain | 703 | 617 | 86 | | 606 | 532 | 75 |
| Sweden | 9 | 6 | 3 | | 11 | 7 | 4 |
| United Kingdom | 36 | 12 | 24 | | 36 | 12 | 24 |
| **EU-28** | **3175** | **2294** | **881** | | **2678** | **1856** | **822** |

### Table 15. Effect of changing income elasticity from 0 to 1.5.

| Country | Elasticity = 0 | Elasticity = 1.0 | Elasticity = 1.5 | Difference |
|---|---|---|---|---|
| | €M/year | €M/year | €M/year | |
| Austria | *46* | 57 | 63 | 11% |
| Belgium | *0* | 0 | 0 | 6% |
| Bulgaria | *101* | 45 | 30 | -33% |
| Croatia | 0 | 0 | 0 | -24% |
| Cyprus | 6 | 5 | 5 | -7% |
| Czech Rep. | 6 | 4 | 4 | -13% |
| Denmark | 13 | 15 | 17 | 9% |
| Estonia | 13 | 9 | 7 | -18% |
| Finland | 5 | 6 | 6 | 4% |
| France | 700 | 720 | 730 | 1% |
| Germany | 329 | 382 | 412 | 8% |
| Greece | 132 | 93 | 78 | -16% |
| Hungary | 84 | 53 | 42 | -20% |
| Ireland | 0 | 0 | 0 | 10% |
| Italy | 252 | 240 | 234 | -2% |
| Latvia | 34 | 21 | 16 | -22% |
| Lithuania | 43 | 29 | 24 | -18% |
| Luxembourg | 3 | 6 | 10 | 58% |
| Malta | | | | |
| Netherlands | 31 | 38 | 41 | 10% |
| Poland | *305* | 193 | 153 | -20% |
| Portugal | 74 | 53 | 45 | -15% |
| Romania | 163 | 77 | 53 | -31% |
| Slovakia | 75 | 54 | 45 | -15% |
| Slovenia | 11 | 8 | 7.44 | -11% |
| Spain | 703 | 637 | 606 | -5% |
| Sweden | 9 | 10 | 11 | 9% |
| United Kingdom | *36* | 36 | 36 | 0% |
| **EU-28** | **3175** | **2,793** | **2678** | -4% |

### 4.3.6 Additional opportunity for adjustment

It is acknowledged that alternative options for adjustment are possible. For example, one could apply an elasticity were the WTP declines as the amount of natural capital per head of population in a country increases. It would clearly be illogical for a country with a small population within an enormous area to value each hectare at the same level as for a country with a very large population contained within a very small area.

The Natura 2000 network is clearly artificial in its limits. Consideration therefore needs to go also to other ecosystems. One option for sensitivity analysis could be to value areas outside the network at 50% less than Natura 2000 areas.

# 5 Repair cost approach

## 5.1 Methods and data

This approach is based on the position that policy inaction (a failure to reduce emissions) would increase restoration costs. This analysis multiplies the restoration cost/damage estimates generated by Ott et al by emissions in each country (see Table 4 above, and Figure 8 and Figure 9). For the purpose of illustration, it is assumed that emission = deposition (what goes up must come down: though an obvious refinement would be to use average deposition rates at the national level to derive an estimate of damage occurring in each country, rather than damage caused by each country). It is also assumed that the correct emission estimate to use with the Ott damage estimates is the mass of NOx and $NH_3$ emitted, rather than the mass emission of nitrogen per se.

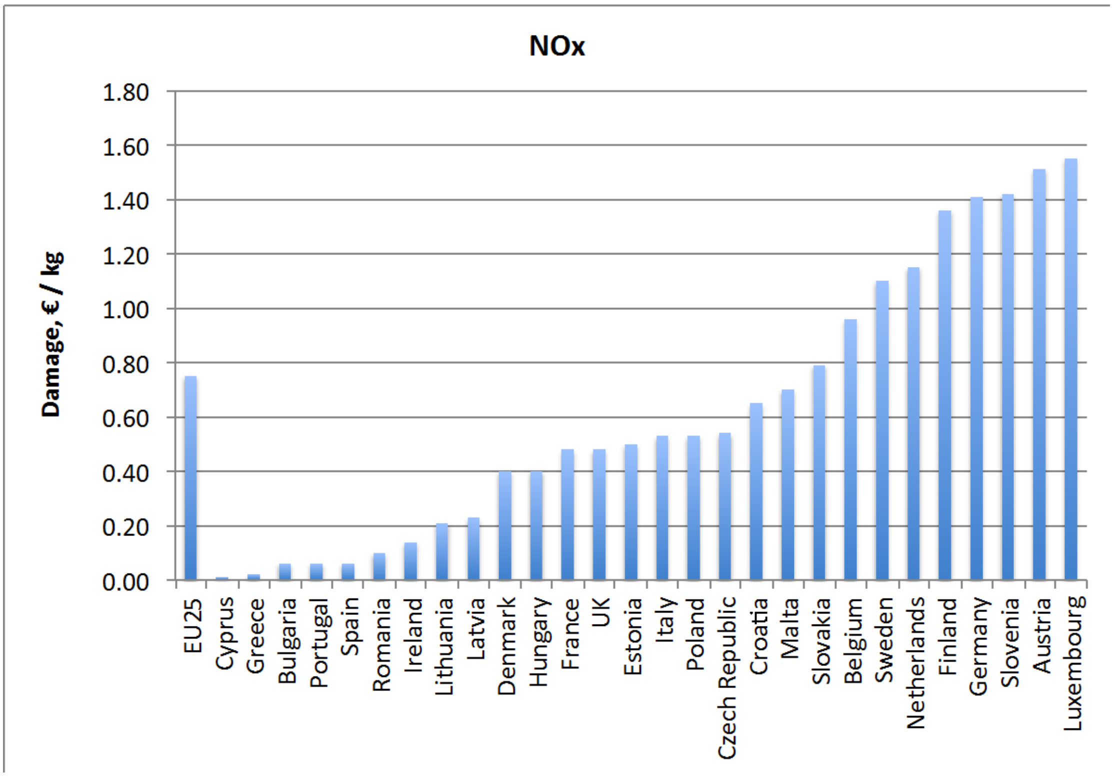



**Figure 8. Ranked externality estimates for NOx (data source: Ott et al, 2006).**

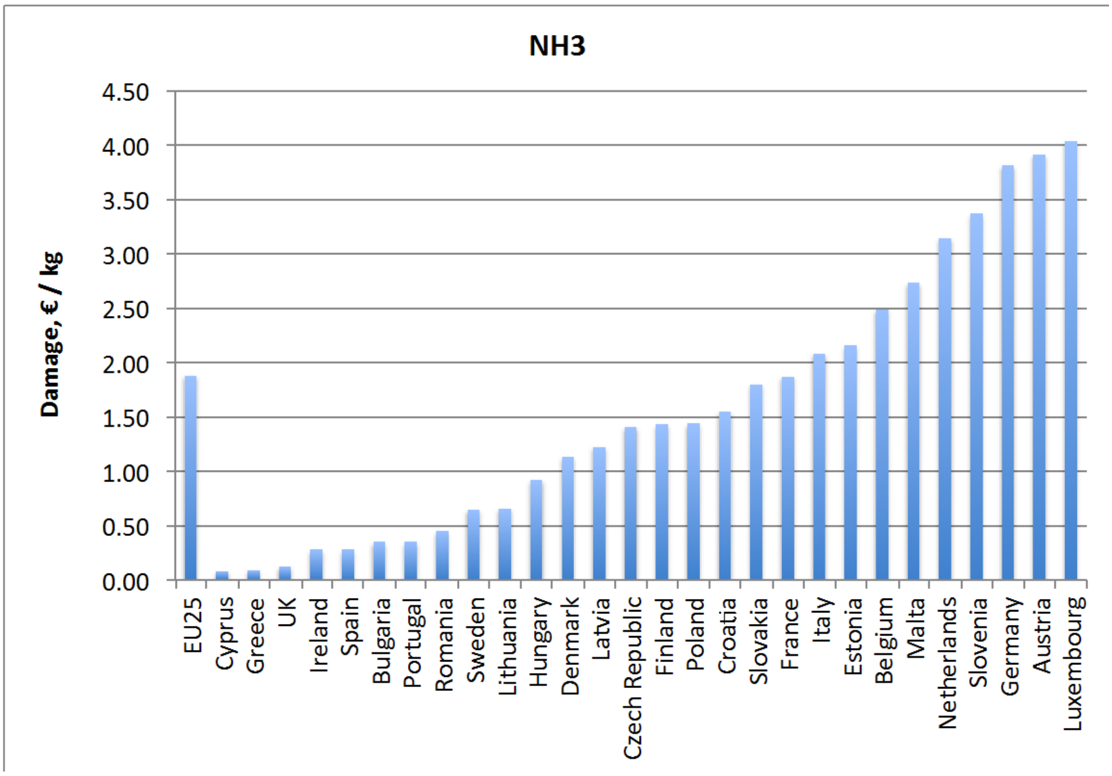



**Figure 9. Ranked externality estimates for $NH_3$ (data source: Ott et al, 2006).**

## 5.2 Results: Estimated damage caused by each country

Results are shown for NOx (Table 16), $NH_3$ (Table 17) and NOx and $NH_3$ combined (Table 18). Roughly twice as much damage is linked to $NH_3$ as NOx[11], with benefits (the difference between scenarios) three times higher for $NH_3$. Total damage for the two pollutants combined at the EU28 level is of the order of €9 billion/year under current legislation (CLE) in both 2025 and 2030, falling to €6 billion/year under MTFR, with benefits of €2.7 billion/year, these being dominated by $NH_3$.

As noted above, there are a number of uncertainties involved in the application of the results of Ott et al. These include:

- The results from Ott et al were calculated for a 2004 scenario, since when emissions have fallen.
- There is an assumption that for the purposes of this analysis, deposition is equivalent to emission.
- It is assumed that the damage costs are assessed against mass emission of $NH_3$ and NOx, rather than emission of N alone.

[11] The difference would be substantially greater if it was assumed that the Ott et al results should be applied per unit of nitrogen rather than NOx and $NH_3$.

Against these caveats it is of course to be remembered that the analysis is performed to test different methods and to see whether they provide consistent answers.

### Table 16. Estimated damage to ecosystems under the CLE and MTFR scenarios for 2025 and 2030 for NOx, and benefits of applying MTFR.

| | NOx cost, M€/kt | CLE damage, €M | MTFR damage, €M | Benefit, €M | | CLE damage, €M | MTFR damage, €M | Benefit, €M |
|---|---|---|---|---|---|---|---|---|
| | | 2025 | 2025 | 2025 | | 2030 | 2030 | 2030 |
| Austria | 1.51 | 116 | 98 | 18 | | 99 | 82 | 17 |
| Belgium | 0.96 | 140 | 106 | 34 | | 129 | 91 | 38 |
| Bulgaria | 0.06 | 4 | 3 | 1 | | 4 | 2 | 1 |
| Croatia | 0.65 | 23 | 11 | 13 | | 22 | 9 | 12 |
| Cyprus | 0.01 | 0 | 0 | 0 | | 0 | 0 | 0 |
| Czech Republic | 0.54 | 70 | 53 | 17 | | 60 | 45 | 16 |
| Denmark | 0.4 | 28 | 22 | 6 | | 25 | 18 | 6 |
| Estonia | 0.5 | 9 | 6 | 3 | | 8 | 5 | 3 |
| Finland | 1.36 | 150 | 126 | 25 | | 135 | 111 | 24 |
| France | 0.48 | 241 | 189 | 52 | | 212 | 159 | 53 |
| Germany | 1.41 | 857 | 648 | 209 | | 748 | 536 | 211 |
| Greece | 0.02 | 3 | 2 | 1 | | 3 | 2 | 1 |
| Hungary | 0.4 | 24 | 17 | 7 | | 21 | 14 | 7 |
| Ireland | 0.14 | 9 | 7 | 2 | | 6 | 4 | 2 |
| Italy | 0.53 | 272 | 221 | 51 | | 242 | 191 | 51 |
| Latvia | 0.23 | 5 | 4 | 1 | | 5 | 3 | 1 |
| Lithuania | 0.21 | 6 | 5 | 1 | | 6 | 5 | 1 |
| Luxembourg | 1.55 | 20 | 19 | 1 | | 16 | 14 | 1 |
| Malta | 0.7 | 1 | 1 | 0 | | 1 | 1 | 0 |
| Netherlands | 1.15 | 182 | 137 | 44 | | 165 | 120 | 44 |
| Poland | 0.53 | 232 | 182 | 50 | | 201 | 148 | 52 |
| Portugal | 0.06 | 6 | 4 | 2 | | 6 | 3 | 2 |
| Romania | 0.1 | 14 | 9 | 5 | | 13 | 8 | 5 |
| Slovakia | 0.79 | 39 | 28 | 12 | | 37 | 25 | 12 |
| Slovenia | 1.42 | 26 | 22 | 4 | | 22 | 17 | 5 |
| Spain | 0.06 | 30 | 22 | 8 | | 26 | 18 | 8 |
| Sweden | 1.1 | 91 | 79 | 12 | | 83 | 70 | 13 |
| United Kingdom | 0.48 | 242 | 183 | 59 | | 212 | 152 | 60 |
| EU28 | | 2,841 | 2,204 | 638 | | 2,502 | 1,855 | 646 |

## Table 17. Estimated damage to ecosystems under the CLE and MTFR scenarios for 2025 and 2030 for $NH_3$, and benefits of applying MTFR.

| | NH3 cost, M€/kt | CLE damage, €M | MTFR damage, €M | Benefit, €M | | CLE damage, €M | MTFR damage, €M | Benefit, €M |
|---|---|---|---|---|---|---|---|---|
| | | 2025 | 2025 | 2025 | | 2030 | 2030 | 2030 |
| Austria | 3.91 | 263 | 181 | 82 | | 265 | 183 | 83 |
| Belgium | 2.49 | 185 | 150 | 34 | | 183 | 150 | 33 |
| Bulgaria | 0.35 | 22 | 20 | 3 | | 23 | 20 | 3 |
| Croatia | 1.55 | 45 | 28 | 17 | | 46 | 29 | 17 |
| Cyprus | 0.08 | 0 | 0 | 0 | | 0 | 0 | 0 |
| Czech Republic | 1.41 | 89 | 73 | 17 | | 88 | 72 | 16 |
| Denmark | 1.13 | 57 | 44 | 13 | | 57 | 44 | 13 |
| Estonia | 2.16 | 27 | 18 | 9 | | 28 | 18 | 10 |
| Finland | 1.43 | 45 | 35 | 10 | | 45 | 35 | 10 |
| France | 1.87 | 1,193 | 795 | 398 | | 1,195 | 794 | 401 |
| Germany | 3.81 | 2,173 | 1,139 | 1,034 | | 2,152 | 1,119 | 1,034 |
| Greece | 0.09 | 4 | 3 | 1 | | 4 | 3 | 1 |
| Hungary | 0.92 | 62 | 44 | 18 | | 62 | 45 | 18 |
| Ireland | 0.28 | 28 | 24 | 4 | | 28 | 24 | 4 |
| Italy | 2.08 | 802 | 615 | 187 | | 810 | 623 | 187 |
| Latvia | 1.22 | 18 | 15 | 3 | | 19 | 15 | 3 |
| Lithuania | 0.66 | 33 | 21 | 12 | | 33 | 21 | 12 |
| Luxembourg | 4.03 | 22 | 18 | 4 | | 22 | 18 | 4 |
| Malta | 2.73 | 4 | 3 | 1 | | 4 | 3 | 1 |
| Netherlands | 3.14 | 351 | 346 | 6 | | 347 | 342 | 5 |
| Poland | 1.44 | 477 | 327 | 149 | | 479 | 329 | 150 |
| Portugal | 0.35 | 25 | 17 | 8 | | 25 | 18 | 8 |
| Romania | 0.45 | 64 | 50 | 13 | | 64 | 50 | 13 |
| Slovakia | 1.8 | 43 | 30 | 13 | | 43 | 30 | 13 |
| Slovenia | 3.37 | 56 | 46 | 10 | | 56 | 46 | 10 |
| Spain | 0.28 | 98 | 59 | 39 | | 98 | 59 | 39 |
| Sweden | 0.65 | 31 | 25 | 6 | | 32 | 26 | 6 |
| United Kingdom | 0.12 | 34 | 28 | 6 | | 34 | 29 | 6 |
| EU28 | | 6,254 | 4,157 | 2,097 | | 6,243 | 4,143 | 2,100 |

### Table 18. Estimated damage to ecosystems under the CLE and MTFR scenarios for 2025 and 2030 for NOx and $NH_3$ combined, and benefits of applying MTFR.

| | NH3 cost, M€/kt | CLE damage, €M | MTFR damage, €M | Benefit, €M | | CLE damage, €M | MTFR damage, €M | Benefit, €M |
|---|---|---|---|---|---|---|---|---|
| | | 2025 | 2025 | 2025 | | 2030 | 2030 | 2030 |
| Austria | 3.91 | 378 | 279 | 99 | | 364 | 265 | 99 |
| Belgium | 2.49 | 325 | 257 | 69 | | 312 | 241 | 71 |
| Bulgaria | 0.35 | 26 | 23 | 4 | | 26 | 22 | 4 |
| Croatia | 1.55 | 69 | 39 | 30 | | 68 | 38 | 30 |
| Cyprus | 0.08 | 1 | 0 | 0 | | 1 | 0 | 0 |
| Czech Republic | 1.41 | 159 | 126 | 34 | | 148 | 117 | 31 |
| Denmark | 1.13 | 85 | 66 | 19 | | 82 | 62 | 19 |
| Estonia | 2.16 | 36 | 24 | 12 | | 36 | 23 | 12 |
| Finland | 1.43 | 195 | 160 | 35 | | 180 | 146 | 34 |
| France | 1.87 | 1,434 | 984 | 451 | | 1,406 | 953 | 453 |
| Germany | 3.81 | 3,030 | 1,787 | 1,243 | | 2,900 | 1,655 | 1,245 |
| Greece | 0.09 | 7 | 6 | 2 | | 7 | 5 | 2 |
| Hungary | 0.92 | 86 | 61 | 24 | | 83 | 59 | 24 |
| Ireland | 0.28 | 37 | 31 | 6 | | 34 | 28 | 6 |
| Italy | 2.08 | 1,075 | 837 | 238 | | 1,052 | 813 | 238 |
| Latvia | 1.22 | 24 | 19 | 4 | | 23 | 19 | 5 |
| Lithuania | 0.66 | 39 | 26 | 13 | | 39 | 26 | 13 |
| Luxembourg | 4.03 | 42 | 37 | 5 | | 38 | 33 | 5 |
| Malta | 2.73 | 5 | 4 | 1 | | 5 | 4 | 1 |
| Netherlands | 3.14 | 533 | 483 | 50 | | 512 | 462 | 50 |
| Poland | 1.44 | 709 | 509 | 200 | | 680 | 477 | 202 |
| Portugal | 0.35 | 31 | 21 | 10 | | 31 | 21 | 10 |
| Romania | 0.45 | 78 | 60 | 18 | | 76 | 58 | 18 |
| Slovakia | 1.8 | 82 | 57 | 25 | | 80 | 54 | 25 |
| Slovenia | 3.37 | 82 | 68 | 14 | | 78 | 64 | 14 |
| Spain | 0.28 | 128 | 81 | 47 | | 124 | 77 | 47 |
| Sweden | 0.65 | 122 | 104 | 18 | | 115 | 96 | 19 |
| United Kingdom | 0.12 | 276 | 211 | 65 | | 246 | 180 | 66 |
| EU28 | | 9,096 | 6,361 | 2,735 | | 8,745 | 5,999 | 2,746 |

# 6 Regulatory revealed preference approach

## 6.1 Methods and data

This method is based on the assumptions that the costs of the Birds and Habitats Directives provide a minimal societal valuation for the protection of the Natura 2000 network, and that policy makers implicitly factored in necessary air pollutant abatement costs when designing the legislation. The analysis brings together the following data:

- The extent of exceedance of the critical load for eutrophication in each country in Natura 2000 areas (terrestrial only)
- The costs of applying all technical $NH_3$ controls contained in the GAINS model
- The same for NOx
- In the event that the critical load is not met, the cost of further measures.

Analysis is performed for both 2025 and 2030 using data generated by IIASA during the development of the Clean Air Policy Package. The baseline for costs is taken as the Current Legislation (CLE) scenario with the upper bound (so far as the analysis using the GAINS outputs is concerned), the Maximum Technically Feasible Reduction (MTFR) scenario.

A lack of consistency has been detected between the IIASA figures on the area of Natura 2000 sites in each country and the data of the Natura 2000 barometer published by the European Commission[12]. To some extent this may be a result of new Natura 2000 areas being declared recently that have yet to be included in the IIASA dataset. However, this does not fully explain the observed discrepancies as for some countries the IIASA data suggest a larger area of Natura 2000 sites than the barometer. Further discussion is needed to understand the differences that have been observed. For the purpose of this report these discrepancies have been ignored in order to demonstrate the analysis.

## 6.2 Results

Data on the geographic extent of the exceedance of the critical load for eutrophication in Natura 2000 areas is presented in Figure 10. This shows some exceedance in all countries except Malta, for which no data were provided. The IIASA data provided no information for 5 of the countries for which figures are presented: Austria, Belgium, Bulgaria, Poland and the UK. However, whilst no information was available for these countries with respect to the Natura 2000

[12] http://ec.europa.eu/environment/nature/natura2000/barometer/index_en.htm

sites, it was available for exceedance across all terrestrial ecosystems. For the purposes of the figure it is assumed that for the countries omitted by IIASA there are proportionally similar levels of exceedance in Natura 2000 sites and all ecosystems. This assumption is largely ratified by inspection of data for countries for which both sets of results were available.

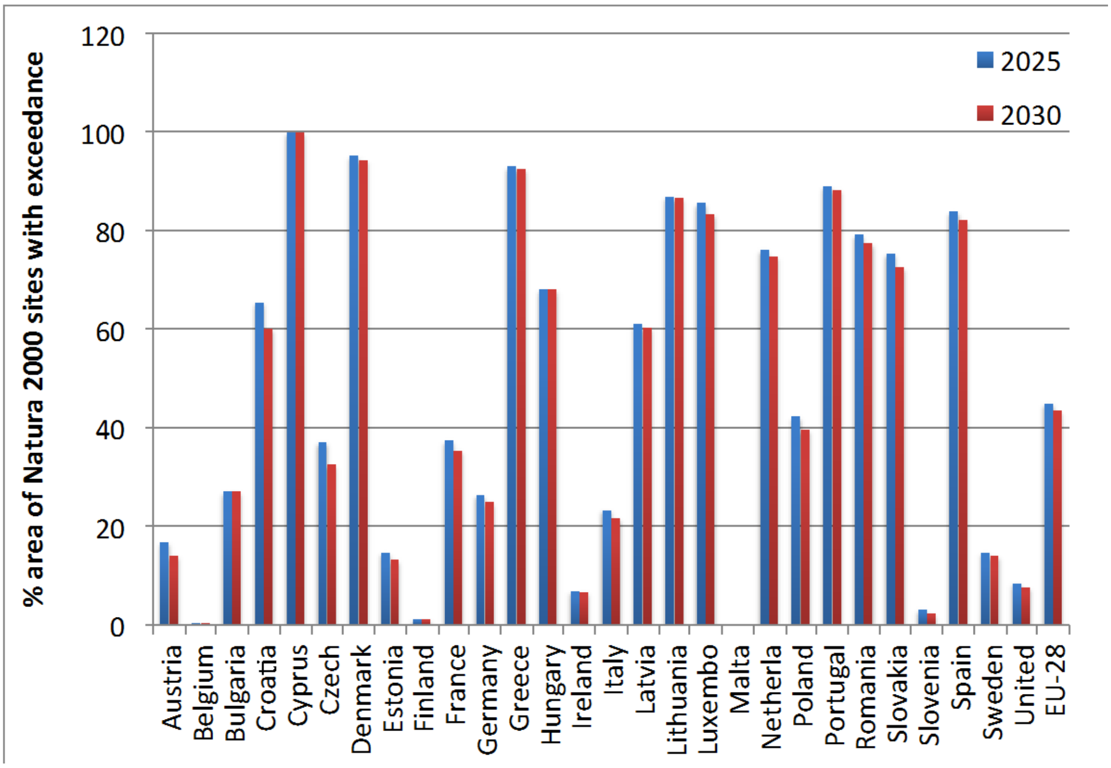



**Figure 10. % area of terrestrial Natura 2000 sites with exceedance of the critical load for eutrophication in 2025 and 2030 under the MTFR scenario.**

There are some possible discrepancies in the results shown in Figure 10, with large differences in levels of exceedance between neighbouring countries. For example, in Belgium there is almost no exceedance, whilst in the Netherlands there is exceedance across more than 70% of the Natura 2000 area. This has further consequences, discussed at the end of this section.

Data on costs of moving from current legislation are shown in Table 19 and Table 20. The much higher costs under CLE for NOx compared to $NH_3$ are a function of higher levels of existing controls. The costs of moving from CLE to MTFR for ammonia are nearly identical for both years, around €4 billion/year. However, the costs of moving to MTFR for NOx are higher for 2030 than 2025 (€6.6 billion against €5.2 billion/year). Total costs for the two pollutants for this shift are in the region of €10 billion for each year.

**Table 19. Costs of moving from the Current Legislation scenario to the Maximum Technically Feasible Reduction (MTFR) scenario in 2025. Data source: IIASA, units €million/year.**

| | CLE $NO_x$, 2025 | MTFR $NO_x$, 2025 | CLE $NH_3$, 2025 | MTFR $NH_3$, 2025 | Change NOx 2025 | Change NH3 2025 | Total cost $NO_x + NH_3$ |
|---|---|---|---|---|---|---|---|
| Austria | 1,272 | 1,415 | 15 | 111 | 143 | 96 | 239 |
| Belgium | 1,292 | 1,454 | 79 | 204 | 162 | 125 | 287 |
| Bulgaria | 731 | 804 | 8 | 43 | 73 | 35 | 108 |
| Croatia | 223 | 304 | - | 45 | 81 | 45 | 126 |
| Cyprus | 99 | 110 | 5 | 15 | 11 | 10 | 21 |
| Czech Rep. | 988 | 1,130 | 28 | 62 | 142 | 34 | 176 |
| Denmark | 570 | 635 | 178 | 248 | 65 | 71 | 136 |
| Estonia | 120 | 140 | 3 | 19 | 20 | 16 | 36 |
| Finland | 727 | 844 | 16 | 64 | 117 | 49 | 166 |
| France | 6,262 | 7,042 | 98 | 737 | 780 | 639 | 1,419 |
| Germany | 7,716 | 8,232 | 158 | 1,158 | 516 | 1,000 | 1,515 |
| Greece | 1,558 | 1,699 | 5 | 34 | 142 | 28 | 170 |
| Hungary | 679 | 782 | 21 | 75 | 102 | 53 | 156 |
| Ireland | 689 | 745 | 30 | 75 | 56 | 45 | 100 |
| Italy | 6,726 | 7,189 | 131 | 419 | 463 | 287 | 750 |
| Latvia | 259 | 286 | 3 | 17 | 27 | 14 | 41 |
| Lithuania | 274 | 312 | 7 | 100 | 37 | 93 | 130 |
| Luxembourg | 107 | 113 | - | 4 | 6 | 4 | 10 |
| Malta | 93 | 96 | - | 2 | 2 | 2 | 4 |
| Netherlands | 2,136 | 2,325 | 398 | 411 | 190 | 12 | 202 |
| Poland | 6,244 | 6,784 | 90 | 506 | 540 | 416 | 956 |
| Portugal | 986 | 1,081 | 13 | 86 | 96 | 73 | 168 |
| Romania | 1,303 | 1,483 | 24 | 132 | 180 | 108 | 288 |
| Slovakia | 528 | 596 | 6 | 30 | 68 | 24 | 92 |
| Slovenia | 273 | 293 | 5 | 16 | 20 | 11 | 30 |
| Spain | 4,995 | 5,507 | 306 | 828 | 511 | 522 | 1,033 |
| Sweden | 836 | 890 | 14 | 65 | 54 | 50 | 104 |
| UK | 5,358 | 5,990 | 67 | 201 | 632 | 135 | 767 |
| **EU-28** | **53,046** | **58,282** | **1,708** | **5,703** | **5,236** | **3,995** | **9,231** |

**Table 20. Costs of moving from the Current Legislation scenario to the Maximum Technically Feasible Reduction (MTFR) scenario in 2030. Data source: IIASA, units €million/year.**

| | CLE $NO_x$, 2025 | MTFR $NO_x$, 2025 | CLE $NH_3$, 2025 | MTFR $NH_3$, 2025 | Change NOx 2025 | Change NH3 2025 | Total cost $NO_x + NH_3$ |
|---|---|---|---|---|---|---|---|
| Austria | 1,383 | 1,575 | 15 | 111 | 192 | 96 | 288 |
| Belgium | 1,428 | 1,592 | 77 | 199 | 164 | 122 | 286 |
| Bulgaria | 743 | 835 | 8 | 44 | 91 | 35 | 127 |
| Croatia | 239 | 322 | - | 46 | 82 | 46 | 128 |
| Cyprus | 114 | 126 | 5 | 16 | 12 | 10 | 23 |
| Czech Rep. | 1,086 | 1,239 | 28 | 62 | 153 | 33 | 187 |
| Denmark | 615 | 702 | 177 | 247 | 87 | 70 | 157 |
| Estonia | 131 | 151 | 3 | 20 | 20 | 17 | 37 |
| Finland | 781 | 910 | 16 | 64 | 129 | 49 | 177 |
| France | 6,843 | 7,688 | 99 | 739 | 845 | 641 | 1,485 |
| Germany | 7,747 | 8,276 | 158 | 1,150 | 528 | 992 | 1,520 |
| Greece | 1,672 | 1,890 | 5 | 34 | 217 | 29 | 246 |
| Hungary | 754 | 861 | 22 | 75 | 108 | 54 | 161 |
| Ireland | 852 | 922 | 31 | 76 | 70 | 45 | 114 |
| Italy | 7,326 | 7,872 | 132 | 423 | 546 | 290 | 836 |
| Latvia | 291 | 328 | 3 | 18 | 37 | 15 | 51 |
| Lithuania | 310 | 353 | 7 | 100 | 43 | 93 | 136 |
| Luxembourg | 114 | 120 | - | 4 | 6 | 4 | 11 |
| Malta | 99 | 101 | - | 2 | 2 | 2 | 4 |
| Netherlands | 5,330 | 6,037 | 388 | 401 | 707 | 12 | 719 |
| Poland | 6,662 | 7,391 | 95 | 520 | 729 | 426 | 1,155 |
| Portugal | 1,114 | 1,220 | 14 | 88 | 106 | 75 | 180 |
| Romania | 1,436 | 1,635 | 24 | 131 | 199 | 107 | 306 |
| Slovakia | 595 | 665 | 6 | 30 | 70 | 25 | 94 |
| Slovenia | 295 | 321 | 5 | 16 | 25 | 11 | 36 |
| Spain | 5,844 | 6,465 | 311 | 837 | 621 | 526 | 1,146 |
| Sweden | 870 | 946 | 15 | 65 | 75 | 51 | 126 |
| UK | 5,597 | 6,335 | 67 | 203 | 738 | 136 | 875 |
| **EU-28** | **60,275** | **66,877** | **1,710** | **5,720** | **6,602** | **4,010** | **10,612** |

Figure 11 investigates the relationship between costs and the area subject to exceedance in each country. There is clearly some correlation between the two: big countries will tend to have larger areas of Natura 2000 sites and more polluting activity. However, there are several factors that weaken the relationship, for example relating to variation in the criteria for declaration of Natura 2000 sites at the national level, variability in ecosystems and their sensitivity to nitrogen and levels of economic activity per unit land area.

Considering the results of Figure 11, it is concluded that a relationship is present, but not very strong. An alternative approach is to allocate out the total EU cost of moving from CLE to MTFR according to the area of exceedance of Natura 2000 sites in each country under CLE. This approach is applied in Table 21 and Table 22. The consistency for each country between the results shown in these tables, and the results shown in Table 19 and Table 20 is considered in Figure 12. Results are very inconsistent, though this is not surprising given the presence of a significant number of countries with limited extent of exceedance of the critical load for eutrophication but abatement opportunity.

Next, it is necessary to consider whether adoption of the MTFR would be required for both pollutants. Emissions of $NH_3$ will in general be deposited closer to source than emissions of NOx, and hence action to protect Natura 2000 sites might focus on $NH_3$. If that is the case, the NOx costs could be ignored, but more action on $NH_3$ would be needed. At a hypothetical extreme, one might consider closing down livestock production. The following calculations for the Netherlands give an indication of magnitudes:

- Value added of livestock farming = 40% of value added of agricultural sector
- Share of agricultural sector in total GDP in the Netherlands = 1.6%
- Total GDP in the Netherlands = €600bn
- Cost of closing livestock units in the Netherlands = €3.8bn

The Netherlands provides an extreme example, given the very extensive exceedance of the critical load for eutrophication in the country. However, this simple calculation suggests that, if widespread closure of livestock facilities were to be applied, costs would be very high for countries with significant levels of exceedance.

It is worth noting that, if a shift to the MTFR scenario were agreed in the interests of protecting ecosystems, the implied valuation of ecosystems would be between a factor 4 and 10 higher than the valuation implicitly accepted for health in the development of the Clean Air Policy Package.

There are several ways in which this analysis could be refined:

- Assess only for $NH_3$ if it is particularly dominant in generating critical loads exceedance.
- Include MTFR options only in countries where it is needed to meet critical loads, depending on transboundary considerations.

- Factor in uncertainty in critical load definition to exclude measures with excessive costs.
- Estimation of livestock reduction necessary after the above are taken into account.

Further consideration will be given to these potential refinements once response to the initial results has been addressed.

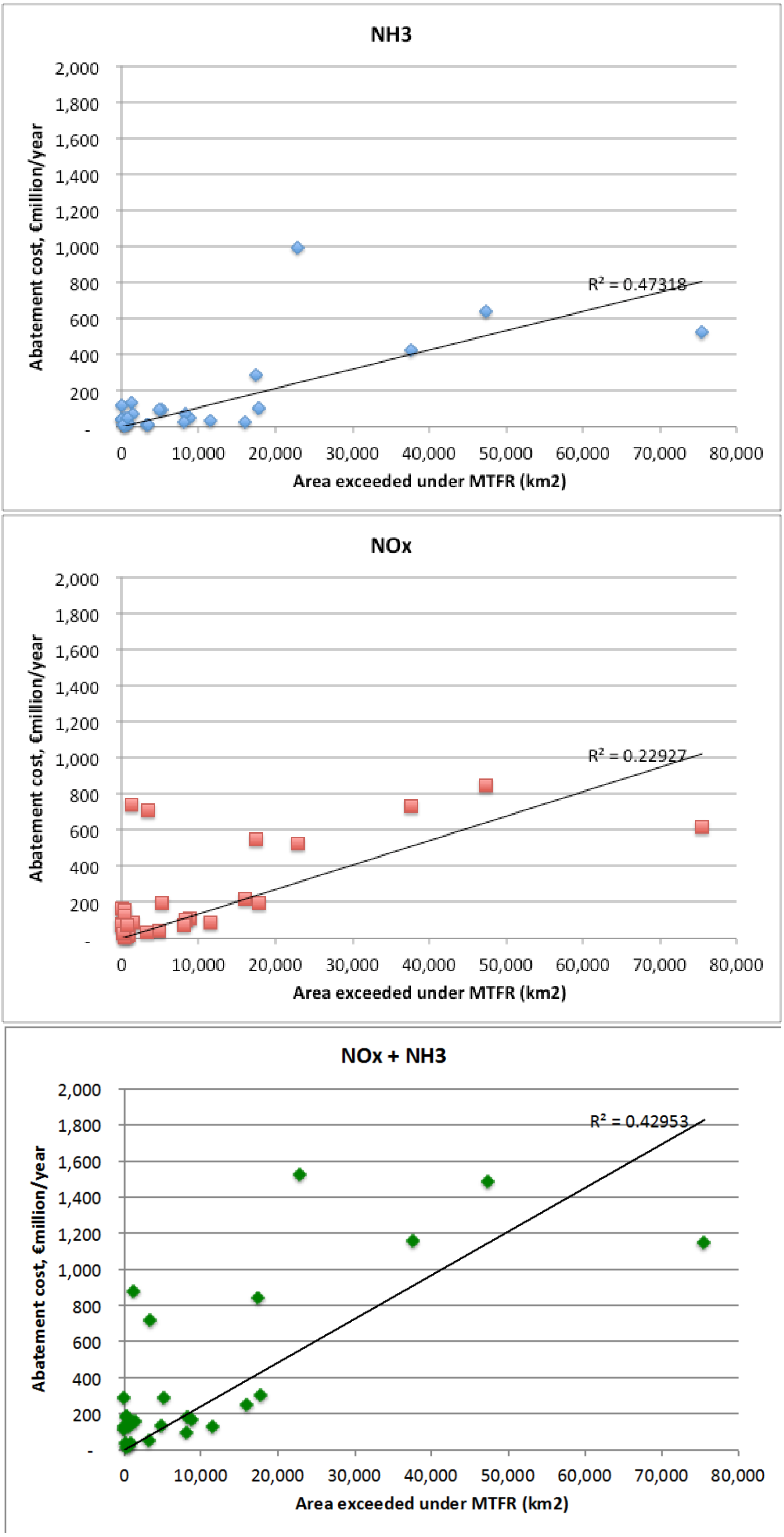



**Figure 11. Relationship between MTFR abatement cost and Natura 2000 area exceeded in each country. Log scale on y-axis.**

**Table 21. Benefit (million/year) to each country of reducing emissions from the CLE scenario in 2025. Shading highlights countries for which data on exceedance of critical loads in Natura 2000 sites was not provided by IIASA.**

| 2025 | **NOx** | **NH3** | **Nox+NH3** |
|---|---|---|---|
| *Cost, € per km2* | *13193* | *10066* | *23259* |
| Austria | 76 | 58 | 134 |
| Belgium | 0 | 0 | 0 |
| Bulgaria | 166 | 127 | 292 |
| Croatia | 0 | 0 | 1 |
| Cyprus | 10 | 8 | 18 |
| Czech Rep. | 9 | 7 | 16 |
| Denmark | 21 | 16 | 37 |
| Estonia | 22 | 17 | 39 |
| Finland | 9 | 7 | 16 |
| France | 1155 | 881 | 2036 |
| Germany | 543 | 414 | 957 |
| Greece | 217 | 166 | 383 |
| Hungary | 139 | 106 | 245 |
| Ireland | 1 | 0 | 1 |
| Italy | 416 | 317 | 733 |
| Latvia | 57 | 43 | 100 |
| Lithuania | 71 | 54 | 125 |
| Luxembourg | 4 | 3 | 7 |
| Malta | | | |
| Netherlands | 51 | 39 | 91 |
| Poland | 504 | 384 | 888 |
| Portugal | 122 | 93 | 215 |
| Romania | 268 | 205 | 473 |
| Slovakia | 123 | 94 | 217 |
| Slovenia | 17 | 13 | 31 |
| Spain | 1160 | 885 | 2045 |
| Sweden | 14 | 11 | 26 |
| United Kingdom | 60 | 46 | 106 |
| EU-28 | 5236 | 3995 | 9231 |

**Table 22. Benefit (million/year) to each country of reducing emissions from the CLE scenario in 2030. Shading highlights countries for which data on exceedance of critical loads in Natura 2000 sites was not provided by IIASA.**

| 2030 | **NOx** | **NH3** | **Nox+NH3** |
|---|---|---|---|
| *Cost, € per km2* | *16953* | *10297* | *27249* |
| Austria | 91 | 55 | 147 |
| Belgium | 0 | 0 | 0 |
| Bulgaria | 213 | 129 | 343 |
| Croatia | 0 | 0 | 1 |
| Cyprus | 13 | 8 | 21 |
| Czech Rep. | 11 | 7 | 18 |
| Denmark | 27 | 16 | 44 |
| Estonia | 28 | 17 | 46 |
| Finland | 11 | 6 | 17 |
| France | 1438 | 873 | 2311 |
| Germany | 679 | 413 | 1092 |
| Greece | 278 | 169 | 447 |
| Hungary | 173 | 105 | 278 |
| Ireland | 1 | 0 | 1 |
| Italy | 514 | 312 | 826 |
| Latvia | 72 | 44 | 115 |
| Lithuania | 91 | 55 | 146 |
| Luxembourg | 5 | 3 | 9 |
| Malta | 0 | 0 | 0 |
| Netherlands | 66 | 40 | 106 |
| Poland | 638 | 387 | 1025 |
| Portugal | 156 | 95 | 251 |
| Romania | 343 | 208 | 551 |
| Slovakia | 156 | 95 | 251 |
| Slovenia | 20 | 12 | 32 |
| Spain | 1485 | 902 | 2387 |
| Sweden | 18 | 11 | 29 |
| United Kingdom | 75 | 45 | 120 |
| EU-28 | 6602 | 4010 | 10612 |

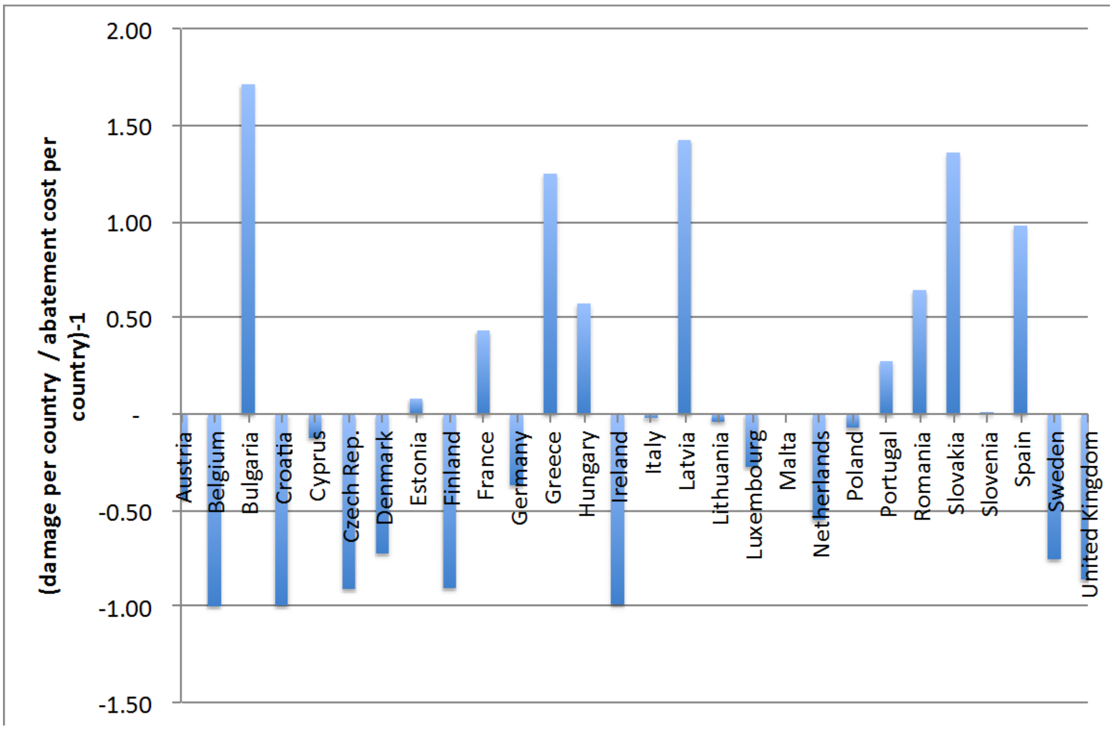



**Figure 12. Test of consistency for abatement cost per country and damage cost per country based on overall EU28 abatement cost using data for 2025**

# 7 Crop and forest damage

## 7.1 General approach for crop impact assessment

In recent years, experts in assessment of crop damage have developed a strong preference for quantification using a measure of dose (referred to as $POD_y$, the phytotoxic ozone dose in excess of some threshold *y*) rather than a measure of concentration (generally expressed as 'AOT40', the accumulated exposure to ozone in excess of 40 ppb over the growing season). The dose metric is preferred as high-ozone conditions with high temperatures may lead to low ozone uptake given limited water availability. Unfortunately, POD relationships are available for very few crops – wheat, tomato and potato at the present time. In order to gain an understanding of the overall effect on crop production it is therefore necessary to make some judgement of the relative sensitivity of a large number of crops compared to those for which POD data are available. This is explored below.

Climate change will cause farmers to switch to different crops that can withstand altered conditions with respect to rainfall, temperature, etc. This is not taken into account below but is being considered by the University of Aarhus.

Analysis is not presented for forest damage, though the methods used are the same, basing production data on FAO statistics and using dose based response functions.

## 7.2 Methods for crop impact assessment

Analysis proceeds through the following steps:
Step 1: Obtain crop production data as economic value of production
Step 2: Convert production data from international $ to euro
Step 3: Define response functions
Step 4: Define geographic resolution
Step 5: Obtain ozone data
Step 6: Apply response functions and calculate impacts

Each step is described below.

**Step 1: Obtain crop production data**
European crop production data for 2010 was extracted from the UN Food and Agriculture Organization (FAO) as gross production, 000 $int[13]. The response functions indicate a linear relationship between the selected metric of ozone exposure and yield. Assuming that the value of yield loss over the range of possible changes in ozone exposure is also linear, it is possible to use the change in economic production directly. If it is assumed that the value of crops does not vary in a linear fashion with yield over

[13] http://faostat3.fao.org/download/Q/QV/E.

the range of interest it may be necessary to go first through a calculation of the change in yield and then to valuation, but this would be a simple addition to the analysis.

**Step 2: Convert production data to euro**
Int$ 2004-6 are converted to 2005 € using a conversion factor of 0.8912. Note: all cost data in the GAINS and ALPHA-Riskpoll Cost-Benefit Assessment models are expressed in price year 2005.

**Step 3: Define response functions**
Response functions identified for each crop type:

For wheat, relative yield = 1 – POD3IAM * 0.0064 (ICP M&M, 2014)

POD relationships are also available for tomato and potato, but only against the POD6 metric, which differs to POD3IAM with respect to both threshold and the period over which ozone data are assessed (55 days vs 90 days). We take the following functions expressed against POD6 and then pro-rate against the what POD3IAM function:

For wheat, relative yield = 1- POD6 *0.038 (ICP M&M, 2014)
For potato, relative yield = 1- POD6 *0.013 (ICP M&M, 2014)
= 1 – POD3IAM * (0.0064 * 0.013/0.038)
= 1 – POD3IAM * 0.0022
For tomato, relative yield = 1- POD6 *0.0266 (Gonzalez-Fernandez et al)
= 1 – POD3IAM * (0.0064 * 0.0266/0.038)
= 1 – POD3IAM * 0.0045

For a number of other crops information on sensitivity is taken from table 1 (their numbering) of ICP Vegetation (2011):

**Table 1** Grouping of crops by sensitivity of yield to ozone. Values in brackets represent the percentage decrease in yield at a 7h mean ozone concentration of 60 ppb compared to that at 30 ppb.

| Sensitive | Moderately sensitive | Tolerant |
|---|---|---|
| Peas and beans (including peanut) (30) | Alfalfa (14) | Strawberry (1) |
| Sweet potato (28) | Water melon (14) | Oat (-3) |
| Orange (27) | Tomato (13) | Broccoli (-5) |
| Onion (23) | Olive (13) | |
| Turnip (22) | Field mustard (12) | |
| Plum (22) | Sugar beet (11) | |
| Lettuce (19) | Oilseed rape (11) | |
| Wheat (18) | Maize (10) | |
| Soybean (18) | Rice (9) | |
| | Potato (9) | |
| | Barley (6) | |
| | Grape (5) | |

For these crops (excluding wheat, potato and tomato, as POD3IAM functions are derived as above), functions are derived relative to POD3IAM by scaling against wheat yield loss. Hence peas and beans are taken to be 30/18 times as sensitive as wheat, and grape 5/18 times as sensitive. It is considered unlikely that ozone is beneficial to oat or broccoli, so for these crops the response function is set to zero.

For other crops some extrapolation is applied where possible. So, for example, simple cereals such as rye are regarded like oat as being tolerant, and legumes generally are regarded like peas and beans as being highly sensitive.

Other crops not covered by the functions derived so far are taken to have similar sensitivity to grape, the least sensitive of the crops in the 'moderately sensitive' class of the table above. The logic for adopting the function for the least sensitive of the 'moderately sensitive' crops is that experimentation tends to focus on species and cultivars for which a significant response has been observed at some time. A lack of data for a crop might therefore suggest that it is unlikely to be highly sensitive, and hence that it is either tolerant or moderately sensitive. The sensitivity of grape is thus taken as indicative of the break point between the two sensitivity classes.

For sensitivity analysis a low variant has been adopted. Wheat, tomato and potato functions are as above. Crops identified as 'sensitive' are given the same sensitivity as wheat, the least sensitive of the 'sensitive' species in Table 1. Tolerant crops are given a DRF of 0. All other crops are given the same sensitivity as grape, the least sensitive of the 'moderately sensitive' crops in the above table.

**Step 4: Define geographic resolution**
The ozone data provided for scenario analysis in policy development for the European Commission are provided at national level only, though represent a receptor-weighted average for each country. The geographic resolution adopted here is thus the national scale.

**Step 5: Obtain ozone exposure data.**
The following data (see table 2 on next page) were taken from earlier GAINS model runs, carried out in the context of the Clean Air Policy Package issued by the European Commission in December 2013.

Key to the table:
TSAP = Thematic Strategy on Air Pollution
CLE = Current legislation
CLE-OPT = cost optimised version of future CLE scenarios
MTFR = Maximum Technically Feasible Reduction (based on measures included in the GAINS database).

For the purpose of illustration here, the data for 2010 are used to quantify yield under zero ozone (given that the FAO production statistics are also for 2010), and data for 2020 are adopted for analysis as results can be compared directly with those presented for the same year in the 2011 ICP Vegetation report. This is done below. Further analysis assesses the 2030 MTFR (Maximum Technically Feasible Reduction scenario) as an indication of the likely extent of future change in POD3IAM ozone exposure. For the purposes of ECLAIRE we will need to consider scenarios going further into the future, to 2050, but 2030 MTFR provides a useful indication of likely changes.

## Table 23. Phytotoxic ozone dose (mmol/m$^2$) above a threshold of 3 nmol/m$^2$/s - generic crop. Source: GAINS (POD3IAM)

| Country | TSAP-2013 CLE | TSAP-2013 CLE | TSAP-2013 CLE | TSAP-2013 CLE_OPT | TSAP-2013 CLE_OPT | TSAP-2013 MTFR | TSAP-2013 MTFR |
|---|---|---|---|---|---|---|---|
| | 2010 | 2015 | 2020 | 2025 | 2030 | 2025 | 2030 |
| Albania | 12.43 | 11.78 | 11.09 | 10.77 | 10.67 | 10.31 | 10.22 |
| Austria | 18.32 | 16.94 | 15.47 | 14.65 | 14.26 | 13.6 | 13.21 |
| Belarus | 17.16 | 16.42 | 15.74 | 15.37 | 15.26 | 14.91 | 14.81 |
| Belgium | 17.93 | 17.15 | 16.32 | 15.81 | 15.6 | 14.89 | 14.68 |
| Bosnia and Herzegovina | 13.65 | 12.74 | 11.56 | 11.16 | 10.99 | 10.45 | 10.28 |
| Bulgaria | 17.87 | 16.4 | 15.39 | 14.93 | 14.61 | 14.07 | 13.77 |
| Croatia | 18.97 | 17.64 | 16.13 | 15.43 | 15.11 | 14.33 | 14 |
| Cyprus | 13.41 | 13.07 | 12.62 | 12.49 | 12.63 | 12.16 | 12.3 |
| Czech Republic | 18.36 | 17.07 | 15.78 | 15.05 | 14.68 | 14 | 13.64 |
| Denmark | 16.19 | 15.55 | 14.62 | 14.16 | 13.93 | 13.54 | 13.33 |
| Estonia | 15.98 | 15.27 | 14.52 | 14.12 | 13.91 | 13.6 | 13.42 |
| Finland | 12.36 | 11.74 | 11.12 | 10.79 | 10.63 | 10.41 | 10.26 |
| France | 17.42 | 16.41 | 15.23 | 14.59 | 14.29 | 13.8 | 13.52 |
| TFYR Macedonia | 12.65 | 11.79 | 10.83 | 10.42 | 10.22 | 9.98 | 9.78 |
| Germany | 18.02 | 16.96 | 15.8 | 15.13 | 14.83 | 14.24 | 13.95 |
| Greece | 16.36 | 15.44 | 14.58 | 14.07 | 13.85 | 13.43 | 13.24 |
| Hungary | 19.9 | 18.51 | 17.1 | 16.42 | 16.09 | 15.32 | 14.99 |
| Ireland | 7.93 | 7.75 | 7.37 | 7.09 | 6.92 | 6.77 | 6.59 |
| Italy | 20.7 | 19.24 | 17.84 | 17.04 | 16.69 | 15.88 | 15.53 |
| Latvia | 16.41 | 15.67 | 14.9 | 14.46 | 14.27 | 13.94 | 13.75 |
| Lithuania | 17.26 | 16.39 | 15.55 | 15.08 | 14.87 | 14.48 | 14.28 |
| Luxembourg | 17.55 | 16.53 | 15.39 | 14.75 | 14.46 | 13.9 | 13.61 |
| Malta | 21.4 | 20.42 | 19.41 | 18.84 | 18.6 | 17.9 | 17.68 |
| Montenegro | 15.65 | 14.6 | 13.4 | 12.96 | 12.74 | 12.29 | 12.08 |
| Netherlands | 17.54 | 16.88 | 16.13 | 15.67 | 15.47 | 14.8 | 14.62 |
| Norway | 8.31 | 7.99 | 7.53 | 7.28 | 7.16 | 7.05 | 6.94 |
| Poland | 18.54 | 17.41 | 16.28 | 15.68 | 15.39 | 14.79 | 14.5 |
| Portugal | 14.58 | 14.14 | 13.57 | 13.26 | 13.09 | 12.69 | 12.51 |
| Republic of Moldova | 16.61 | 15.88 | 15.14 | 14.84 | 14.8 | 14.34 | 14.3 |
| Romania | 17.47 | 16.4 | 15.42 | 14.94 | 14.73 | 14.05 | 13.84 |
| Russian Federation | 13.05 | 12.55 | 12.15 | 11.9 | 11.87 | 11.73 | 11.71 |
| Serbia | 15.65 | 14.6 | 13.4 | 12.96 | 12.74 | 12.29 | 12.08 |
| Slovakia | 18.54 | 17.26 | 15.95 | 15.29 | 14.97 | 14.19 | 13.86 |
| Slovenia | 18.73 | 17.31 | 15.68 | 14.83 | 14.42 | 13.7 | 13.29 |
| Spain | 11.83 | 11.5 | 10.89 | 10.56 | 10.39 | 10.05 | 9.87 |
| Sweden | 13.41 | 12.82 | 12.04 | 11.66 | 11.49 | 11.22 | 11.05 |
| Switzerland | 11.58 | 10.75 | 9.68 | 9.09 | 8.81 | 8.57 | 8.3 |
| Ukraine | 16.2 | 15.55 | 14.9 | 14.63 | 14.62 | 14.25 | 14.25 |
| United Kingdom | 12.62 | 12.2 | 11.52 | 11.16 | 10.98 | 10.55 | 10.39 |

**Step 6: Application of the response functions.**
Existing production data are of course depressed as a result of exposure to current levels of ozone. A first stage is therefore to quantify a counter-factual level of production, assuming that ozone levels (here, as POD3IAM) = 0. This is calculated for each country using the following expression, where DRF = dose response function:

$$Yield\ at\ zero\ ozone = \frac{2010\ yield}{1 - [POD3IAM \times DRF]}$$

Hence if ambient ozone in 2010 reduced yield of a crop by 20% (the product of POD3IAM and DRF), the yield at zero ozone would have been 25% higher than reported production.

Assuming that crop production patterns remain unchanged, the impacts of ozone in future years can then by calculated as follows:

$Impact_y = 2010\_production * POD3IAM_y * DRF$

Where the subscript *y* refers to the target year for quantification.

## 7.3 Results for crops

Results are shown in Table 24 for each of the crops considered, and Table 25 disaggregated to country. Two of the above series of scenarios are considered, describing impacts in 2010 and forecast effects in 2030 under the Maximum Technically Feasible Reduction Scenario (MTFR), these providing the extremes of the scenarios for which data were available. Overall, the reduction in ozone levels over this period leads to a reduction in annual damage of €1.98 billion.

Results for the EU27 (EU excluding Cyprus) with Norway and Switzerland are shown in Table 26[14]. These results account for 79% of the total for the longer list of European countries. The crop for which the largest damage is estimated is wheat, at over 30% of the total: no other crop provides more than 10% of the total damage. This dominance of wheat is to be expected given that wheat is widely grown and sensitive. The total damage from crops for which POD functions are available (wheat, potato and tomato) is 40%.

[14] This grouping of countries was selected for comparison against results from ICP Vegetation (2011).

## Table 24. Crop loss by crop in 2010 and 2030 under the MTFR scenario.

| All countries | 2010 total production at ambient ozone | 2010 total production at zero ozone | Best estimate 2010 crop loss | Best estimate 2030 MTFR crop loss |
|---|---|---|---|---|
| Wheat | 28,344 | 31,594 | 3,250 | 2,635 |
| Potatoes | 16,198 | 16,784 | 586 | 487 |
| Grapes | 13,643 | 14,064 | 421 | 330 |
| Maize | 10,721 | 11,424 | 702 | 555 |
| Olives | 9,119 | 9,793 | 674 | 539 |
| Barley | 7,787 | 8,056 | 268 | 218 |
| Tomatoes | 7,151 | 7,659 | 508 | 409 |
| Rapeseed | 5,776 | 6,187 | 411 | 325 |
| Sugar beet | 5,766 | 6,163 | 397 | 320 |
| Apples | 5,329 | 5,497 | 169 | 133 |
| Sunflower seed | 4,901 | 5,042 | 141 | 119 |
| Mushrooms and truffles | 2,774 | 2,774 | - | - |
| Peaches and nectarines | 2,063 | 2,126 | 63 | 50 |
| Vegetables, fresh, other | 1,859 | 1,915 | 56 | 45 |
| Strawberries | 1,761 | 1,771 | 10 | 8 |
| Carrots and turnips | 1,746 | 1,995 | 249 | 204 |
| Onions, dry | 1,623 | 1,863 | 240 | 198 |
| Triticale | 1,522 | 1,571 | 49 | 39 |
| Plums and sloes | 1,456 | 1,672 | 215 | 171 |
| Cabbages and other brassicas | 1,426 | 1,468 | 41 | 34 |
| Lettuce and chicory | 1,290 | 1,451 | 161 | 127 |
| Oats | 1,213 | 1,213 | - | - |
| Chillies and peppers, green | 1,205 | 1,239 | 34 | 27 |
| Soybeans | 1,171 | 1,306 | 135 | 112 |
| Oranges | 1,141 | 1,347 | 206 | 163 |
| Rice, paddy | 1,072 | 1,130 | 58 | 47 |
| Pears | 1,050 | 1,082 | 33 | 26 |
| Rye | 1,028 | 1,028 | - | - |
| Almonds, with shell | 983 | 1,010 | 27 | 21 |
| Cucumbers and gherkins | 938 | 964 | 26 | 22 |
| Cherries | 852 | 878 | 26 | 21 |
| Leeks, other alliaceous vegetables | 719 | 742 | 22 | 18 |
| Tangerines, mandarins, clementines, satsumas | 714 | 733 | 19 | 15 |
| Raspberries | 685 | 704 | 19 | 16 |
| Peas, dry | 656 | 785 | 129 | 106 |
| Watermelons | 517 | 560 | 43 | 35 |
| Currants | 512 | 527 | 14 | 12 |
| Cauliflowers and broccoli | 499 | 499 | - | - |
| Grain, mixed | 495 | 495 | - | - |
| Kiwi fruit | 475 | 492 | 17 | 13 |
| Peas, green | 474 | 572 | 98 | 78 |
| Other crops | 10,125 | 10,798 | 673 | 536 |
| **Total** | **158,780** | **168,972** | **10,192** | **8,214** |

## Table 25. Crop loss by country in 2010 and 2030 under the MTFR scenario.

| All crops | 2010 total production at ambient ozone | 2010 total production at zero ozone | Best estimate 2010 crop loss | Low estimate 2030 crop loss |
|---|---|---|---|---|
| Albania | 598 | 625 | 27 | 22 |
| Austria | 1,615 | 1,723 | 107 | 78 |
| Belarus | 3,222 | 3,405 | 183 | 158 |
| Belgium | 2,324 | 2,464 | 140 | 115 |
| Bosnia and Herzegovina | 567 | 597 | 30 | 22 |
| Bulgaria | 2,114 | 2,262 | 148 | 114 |
| Croatia | 845 | 909 | 64 | 47 |
| Czech Republic | 1,581 | 1,719 | 138 | 102 |
| Denmark | 1,682 | 1,810 | 128 | 105 |
| Estonia | 162 | 173 | 11 | 9 |
| Faroe Islands | 0 | 0 | - | - |
| Finland | 582 | 605 | 23 | 19 |
| France | 19,220 | 20,715 | 1,495 | 1,160 |
| Germany | 11,762 | 12,657 | 895 | 693 |
| Greece | 4,930 | 5,242 | 312 | 252 |
| Hungary | 2,963 | 3,198 | 235 | 177 |
| Iceland | 4 | 4 | - | - |
| Ireland | 463 | 474 | 11 | 9 |
| Italy | 18,757 | 20,127 | 1,370 | 1,028 |
| Latvia | 361 | 388 | 27 | 23 |
| Liechtenstein | 0 | 0 | - | - |
| Lithuania | 663 | 716 | 54 | 44 |
| Luxembourg | 37 | 40 | 3 | 2 |
| Malta | 31 | 33 | 2 | 2 |
| Montenegro | 96 | 101 | 5 | 4 |
| Netherlands | 3,391 | 3,588 | 197 | 164 |
| Norway | 248 | 254 | 6 | 5 |
| Poland | 8,845 | 9,391 | 546 | 427 |
| Portugal | 2,115 | 2,225 | 110 | 94 |
| Republic of Moldova | 1,027 | 1,086 | 60 | 52 |
| Romania | 5,401 | 5,788 | 387 | 307 |
| Russian Federation | 18,374 | 19,403 | 1,030 | 924 |
| Serbia | 2,894 | 3,085 | 191 | 147 |
| Slovakia | 633 | 686 | 53 | 39 |
| Slovenia | 243 | 257 | 14 | 10 |
| Spain | 19,319 | 20,186 | 867 | 724 |
| Sweden | 923 | 977 | 53 | 44 |
| Switzerland | 559 | 582 | 23 | 16 |
| The former Yugoslav Republic of Macedonia | 602 | 626 | 24 | 18 |
| Ukraine | 13,843 | 14,696 | 853 | 750 |
| United Kingdom | 5,786 | 6,157 | 371 | 306 |
| **Total** | **158,780** | **168,972** | **10,192** | **8,214** |

## Table 26. Crop loss by crop in 2010 and 2030 under the MTFR scenario in the EU27 (excluding Cyprus) + Norway and Switzerland.

| EU27 (not Cyprus) + NO + CH | Best estimate 2010 crop loss | % of total for 2010 | Best estimate 2030 MTFR crop loss | % of total for 2030 MTFR |
|---|---|---|---|---|
| Wheat | 2,367 | 30.4% | 1,853 | 30.3% |
| Potatoes | 333 | 4.3% | 264 | 4.3% |
| Grapes | 392 | 5.0% | 306 | 5.0% |
| Maize | 513 | 6.6% | 396 | 6.5% |
| Olives | 670 | 8.6% | 536 | 8.8% |
| Barley | 201 | 2.6% | 158 | 2.6% |
| Tomatoes | 407 | 5.2% | 321 | 5.2% |
| Rapeseed | 369 | 4.7% | 289 | 4.7% |
| Sugar beet | 295 | 3.8% | 230 | 3.8% |
| Apples | 136 | 1.7% | 105 | 1.7% |
| Sunflower seed | 55 | 0.7% | 43 | 0.7% |
| Mushrooms and truffles | - | 0.0% | - | 0.0% |
| Peaches and nectarines | 61 | 0.8% | 48 | 0.8% |
| Vegetables, fresh, other | 40 | 0.5% | 32 | 0.5% |
| Strawberries | 8 | 0.1% | 6 | 0.1% |
| Carrots and turnips | 176 | 2.3% | 140 | 2.3% |
| Onions, dry | 165 | 2.1% | 132 | 2.2% |
| Triticale | 43 | 0.6% | 34 | 0.6% |
| Plums and sloes | 140 | 1.8% | 109 | 1.8% |
| Cabbages and other brassicas | 22 | 0.3% | 17 | 0.3% |
| Lettuce and chicory | 160 | 2.1% | 125 | 2.0% |
| Oats | - | 0.0% | - | 0.0% |
| Chillies and peppers, green | 27 | 0.3% | 22 | 0.4% |
| Soybeans | 42 | 0.5% | 32 | 0.5% |
| Oranges | 206 | 2.6% | 162 | 2.7% |
| Rice, paddy | 45 | 0.6% | 35 | 0.6% |
| Pears | 29 | 0.4% | 23 | 0.4% |
| Rye | - | 0.0% | - | 0.0% |
| Almonds, with shell | 27 | 0.3% | 21 | 0.3% |
| Cucumbers and gherkins | 14 | 0.2% | 11 | 0.2% |
| Cherries | 20 | 0.3% | 15 | 0.3% |
| Leeks, other alliaceous vegetables | 22 | 0.3% | 17 | 0.3% |
| Tangerines, mandarins, clementines, satsumas | 19 | 0.2% | 15 | 0.2% |
| Raspberries | 8 | 0.1% | 7 | 0.1% |
| Peas, dry | 75 | 1.0% | 59 | 1.0% |
| Watermelons | 24 | 0.3% | 19 | 0.3% |
| Currants | 8 | 0.1% | 6 | 0.1% |
| Cauliflowers and broccoli | - | 0.0% | - | 0.0% |
| Grain, mixed | - | 0.0% | - | 0.0% |
| Kiwi fruit | 17 | 0.2% | 13 | 0.2% |
| Peas, green | 90 | 1.2% | 71 | 1.2% |
| Other crops | 565 | 7.3% | 444 | 7.3% |
| **Total** | **7,790** | | **6,116** | |

## 7.4 Forest damage

As noted above, the methods for assessment of forest damage are essentially the same as those for agricultural production, starting from the use of monetised data on production from FAO.

A complication for the analysis concerns the variety of products reported by FAO that are linked to forest production, from sawn timber to pulp to charcoal and firewood, and to papers of various types. We have no basis for assessing how a change in forest productivity will differentially affect the different types of forest product. The assumption will therefore be made that all products are affected equally. Sensitivity to this assumption will be tested.

## 7.5 Discussion

The results above can be compared against results from table 2 of ICP Vegetation (2011) for wheat and tomato (other crops were not included in the ICP assessment, as it was restricted to those for which POD functions were available):

**Table 2** Predicted impacts of ozone pollution on wheat and tomato yield and economic value, together with critical level exceedance in EU27+Switzerland+Norway in 2000 and 2020 under the current legislation scenario (NAT scenario). Analysis was conducted on a 50 x 50 km EMEP grid square using crop values in 2000 and an ozone stomatal flux-based risk assessment.

| | Wheat | | Tomato | |
|---|---|---|---|---|
| | 2000 | 2020 | 2000 | 2020 |
| Total production, million t | 133.53 | | 17.68 | |
| Total economic value of wheat in 2000, billion Euro | 15.87 | | 6.85 | |
| Mean % yield loss per grid square | 13.7[1] | 9.07[1] | 9.4[2] | 5.7[2] |
| Total production loss, million t | 26.89 | 16.45 | 2.64 | 1.62 |
| Total economic value loss, billion Euro | 3.20 | 1.96 | 1.02 | 0.63 |
| Percentage of EMEP grid squares exceeding critical level | 84.8[1] | 82.2[1] | 77.8[2] | 51.3[2] |

[1]based on all grid squares with wheat production, [2] based on grid squares with > 1 tonne of production

Despite the inconsistency in the years of assessment, it is clear that results for wheat are in good disagreement (economic loss of €1.96 billion for 2020 according to ICP Vegetation, vs €2.4 to €1.85 billion estimated here for 2030 MTFR and 2010). The ICP Vegetation results for tomato are significantly higher, however, a loss of €630 million from ICP Vegetation compared to a range of €300 to 400 million estimated here. It is thought likely that this difference is a function of different assumptions on irrigation between the different POD estimates used. Ongoing discussions with CEH are investigating whether a correction can be introduced to the analysis.

Sensitivity to different assumptions on crop response was tested, for those crops for which POD functions were unavailable, and estimates were based on extrapolation. Sensitivity was small, in large part due to the dominance of the crops for which POD functions were available (together accounting for 40% of damage).

No account is taken of some interactions here. The interaction with climate through variation in crop distributions is being investigated with the University of Aarhus. No attempt will be made to account for interactions with insect pests, given the lack of recent literature in this field, though it is noted as a potentially significant omission.

The approach developed for forests is similar, based on use of POD functions and linked to FAO production data. It has not been tested here as it needs to be performed in conjunction with analysis of carbon sequestration, methods for which are still in development at the time of writing. Valuation of carbon sequestration is, however, straightforward, drawing on a uniform estimate of damage per tonne carbon sequestrated, which will be based on the estimates adopted by EEA (2014).

# 8 Discussion

The methods defined here will be applied using data from the agreed ECLAIRE scenarios for the final report on this work package. As noted above, results will be provided for the key ecosystem services relating to appreciation of biodiversity, crop production, forest production and carbon sequestration.

The analysis of biodiversity impacts presented here provides some advance over previous work (which is retained, through the 'restoration cost' approach and the 'regulatory revealed preference' approach, though neither is considered as theoretically robust as the estimates based on willingness to pay. The main problem with the WTP estimates is that they are based on a very limited literature. An important feature of the selected literature is that the WTP estimates are directly concerned with realistic levels of improvement, linked to the biodiversity action planning process. We acknowledge a broader literature on the valuation of these ecosystem services, but the outputs of that work are not directly applicable to estimated changes in the pollution burden.

A limitation of the approach taken here for biodiversity is that it is annualised, an approach that works well for traded commodities such as crops, or for goods, like human health where the time between exposure and impact is limited. This is, however, less the case for impacts on biodiversity, where change may take many years to occur, and then take decades or centuries to pass, if it ever does. An accumulated estimate of damage over time may provide a different stimulus for action than an annual estimate.

A further issue for biodiversity concerns the transparency of outputs relative to the ecological changes that are occurring. Mapping exceedance, or providing an anthropocentric estimate of economic damage is useful, but does not communicate the types of change that are occurring. This is not a problem if change is well understood by those making the policy. However, this will not include all relevant decision makers, for example those outside government environment departments in finance ministries, for example.

The fact that the estimates of damage provided here are small relative to results for human health impacts does not mean that these results can simply be ignored. Damage is in the order of many billions of euro annually, and potential improvements of several billion euro annually are possible. A focus only on health provides a bias against maximising the benefits that can be accrued from more broadly targeted policies. Recognising damage more broadly should also widen the appeal of progressive air pollution policies, for example to those working in agriculture, who may presently regard air pollution policy as more threat than opportunity.

Further refinement of methods will be reported in the final report on this work package, for example in relation to impacts of climate change on crop production and how it interacts with ozone damage.

# 9 References

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