Green Accounting in Europe

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Green Accounting in Europe
A Comparative Study, Volume 2
Edited by Anil Markandya and Marialuisa Tamborra

Green Accounting in Europe

A Comparative Study, Volume 2

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INSTITUTIONS PARTICIPATING IN THE PROJECT AND THEIR CONTRIBUTIONS:

Organization	**Contribution**
Bath University, UK	Introduction and Project Objectives (Ch. 1); Developments in Green Accounting (Ch.2); Developments in Valuation (Ch. 5); Defensive Expenditures (Ch. 7); Estimates of Damage Costs from Air Pollution to Human Health, Crops and Materials (Ch. 8); Marginal Cost Estimates of Greenhouse Gas Emissions (Ch. 14); Conclusions and Policy Recommendations (Ch. 16).
Fondazione Eni Enrico Mattei, Italy	Defensive Expenditures (Ch. 7); Estimates of Damage Costs from Air Pollution to Human Health, Crops and Materials (Ch. 8); Valuation of Water (Ch. 12); Environmental Expenditures (Ch. 15); Conclusions and Policy Recommendations (Ch. 16).
Institut für Energiewirtschaft und Rationelle Energieanwendung (IER) – Stuttgart University, Germany	The Methodology for the Estimation of Impacts and Damage Costs Caused by Ambient Air Pollution (Ch. 6); Estimates of Damage Costs from Air Pollution to Human Health, Crops and Materials (Ch. 8);

	Attribution of the Damages to Countries and Economic Sectors of Origin (Ch. 9).
AEA Technology Environment, UK	Developments in Pathway Analysis (Ch. 3); Updates to Exposure–Response Functions (Ch. 4); Developments in Valuation (Ch. 5); Estimates of Damage Costs from Air Pollution to Human Health, Crops and Materials (Ch. 8); Damages to Land (Ch. 13).
Institute for Environmental Studies (IVM), Vrije Universiteit, Amsterdam, The Netherlands	Estimates of Damage Costs from Air Pollution to Human Health, Crops and Materials (Ch. 8); Developments in Estimation of Damages to Crops (Ch. 10); Marginal Cost Estimates of Greenhouse Gas Emissions (Ch. 14).
Institute of Occupational Medicine, UK	Updates to Exposure–Response Functions (Ch. 4); Developments in Valuation (Ch. 5).
Institut für Wirtschaftsforschung – IFO, University of München, Germany	Forest and Ecosystem Damages (Ch. 11).

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Preface

This book presents the results of the project GARP II (Green Accounting Research Project), an EU-funded project (Contract No. ENV4-CT96-0285) finished at the end of 1998. GARP II is the follow-up of a previous research project (GARP I), whose results were published in Markandya and Pavan (1999), *Green Accounting in Europe: Four Case Studies*, Kluwer Academic Publishers).

Both projects used the methodology developed under the series of ExternE projects, known as Impact Pathway Analysis, to calculate damage accounts to be used in a green accounting framework. GARP II broadens the scope of pollutants analysed and – more importantly – goes beyond the case study approach used in GARP I where country results were presented: it illustrates results in a comparative way across different countries. Moreover, results here are attributed to sources of pollution, so that damages can be reported not only in aggregate but also in terms of where they originate from. This allows one to draw important conclusions on sectoral policies. This volume therefore can be considered as a second volume that follows the first publication by Markandya and Pavan (1999).

The ideas raised in this book are refined and explored further in a follow-up project, called GREENSENSE, that looks not only at welfare effects, but also at sustainability issues.

Completion of this book took longer than expected mainly because both of us were called by new challenges: Anil was appointed as advisor at the World Bank in Washington and Marialuisa moved to the European Commission – Directorate General Research. However, the work reflects the ideas and research activity that were undertaken when Anil was at the Department of Economics and International Development at Bath University and scientific co-ordinator of the GARP II project and Marialuisa was researcher and research area co-ordinator at Fondazione Eni Enrico Mattei.

We are grateful to all those colleagues who made the achievement of this book possible; in particular Alistair Hunt, Tim Taylor and Nick Dale of Bath University for their editing and revisions, Carlo Carraro, Research Director at Fondazione Eni Enrico Mattei, for

his support and the reviewers of the first manuscript for their valuable comments.

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PART I

Developments in Methodology

INTRODUCTION

Chapter 1 presents an introduction to the book and highlights its objectives. It also presents the methodological framework. Chapter 2 presents developments that have been made in the area of green accounting since Markandya and Pavan (1999), *Green Accounting in Europe: Four Case Studies* (eds) (Kluwer Academic Publishers) was written. Chapter 3 reviews developments that have been made in the analysis of impact pathways, with Chapters 4 and 5 evaluating the progress made in the estimation of dose-response functions and in the valuation of environmental damages. Chapters 6 and 7 report the methodologies utilised for the estimation of damages caused by ambient air pollution and the costs of defensive expenditures respectively. Part II of this book then presents the results of the analysis and Part III reviews the conclusions that can be drawn for environmental accounting and its use for policy, especially in the EU.

1. Introduction and project objectives

Anil Markandya and Marialuisa Tamborra

1.1 THE CONTEXT

As countries become richer, a heightened interest in the environment comes from two sources. Individuals have a greater concern for the quality of the ambient environment, as more pressing needs are satisfied. At the same time the pressures on that ambient environment increase, with a higher loading of pollution from transport, power, industry and household consumption. One way in which this increased awareness of environmental problems manifests itself is through the demand for better information on what is happening to the environment and what that means for us as citizens and human beings. It is not surprising therefore that there has been an explosion of work on measuring impacts in terms of the *pressures* on the ambient environment, the *state* of that environment and the *responses* of society to these pressures (Adriaanse, 1993).

This work has been carried out almost exclusively in physical units, with little attention paid to the economic implications of the environmental changes. At the same time, it is clear that there is an economic dimension to the changes. The environment provides an economic function and it is at our peril that we ignore that function. But, in drawing up traditional measures of economic activity, such as Gross Domestic Product, that is precisely what we do. We do not take account of damages done to the stock of natural capital, nor of the losses of welfare that economic activities cause through increased pollution.

In response to these concerns, a literature has developed on the monetary value of environmental changes caused by economic activity. This literature has several strands. One looks at the depletion in mineral and renewable resources and asks whether conventional measures of GDP have paid enough attention to this depletion. If a country is maintaining its level of economic activity by running down its mineral resources but is not fully replacing them with alternative income generating assets, then its present level of welfare may be unsustainable. A second strand examines the expenditures undertaken by citizens in protecting themselves from the consequences of increased pollution. These so-called 'defensive expenditures' are subject to some controversy. Should we deduct them from measures of national income? Some argue that if money is spent on such items, and it used to be spent on things that directly gave welfare, then society is indeed worse off. The problems lie in knowing what the relevant point of comparison is, and identifying and measuring these expenditures. Nevertheless this is an important area of work and much remains to be done to achieve satisfactory systems of accounting for defensive expenditures. The third area of work relates to the damage caused by the pollution. Can that damage be measured in money terms? If so, how much is it worth? And how do the values of the damage compare to other measures of economic activity, such as GDP (gross domestic product) or National Wealth?

This book presents the results of a two-phase project and deals with environmental accounting within an economic framework, focusing on the second and third issues raised above, but specially the third. Its point of departure is a project called GARP I, a study by researchers in Germany, Italy, the Netherlands and the UK, and published as a book (Markandya and Pavan, 1999).[1] A summary was published by the UK Office of National Statistics in 1998 (Markandya and Milborrow, 1998). Essentially what this research attempted was to use spatially disaggregated data on measures of pollution to derive economic damage estimates for those pollutants. The objective was to see what could (credibly) be done at a national level, what was the degree of uncertainty in the estimates and whether it was possible to make inter-country comparisons of damages within the European Union. The project also looked at whether these measures could be constructed on a routine basis, so that the task of preparing them could be handed over to statistical offices.

GARP I (Markandya and Pavan, 1999) considered only airborne pollutants, where a spatially disaggregated analysis was undertaken focusing on different *receptors* (such as human health, crops, the built and natural environment). Inevitably, the coverage and approach were not fully consistent. The degree of spatial disaggregation varied by country, as did the availability of data on pollution concentrations and estimates of stocks at risk. These differences accounted for a significant part of the differences in the resulting damage estimates.

In spite of these difficulties, the experience of the first phase confirmed that it is possible to make monetary estimates of the impacts of pollutants on human health, materials and crops with some credibility, although considerable further work is required before these tasks can be carried out routinely and in a comparable format for all countries. Health impacts accounted for the greatest damages, particularly chronic mortality from exposure to particulate pollution. Assessing damages incurred by forests and ecosystems proved much more difficult due to both a lack of appropriate data and the underlying complexity of the natural systems. Some estimates of global warming damages were presented that reflected the limited consensus among certain sections of the scientific research community. However, these figures were regarded as highly uncertain and possibly subject to major revisions. The initial estimates were presented in numerous academic and policy-making fora and stimulated considerable discussion. Although several researchers are convinced of the usefulness of this exercise, it should be noted that there are still many commentators who are strongly opposed to this kind of valuation exercise, particularly when expressing life expectancy and other health issues in money terms.

In GARP I it was concluded that, for this kind of exercise, a hybrid approach is appropriate, where monetary values and physical impacts are presented, with the latter being in the form that is closest to the 'end point' that affects human welfare. The project team also agreed that the data are more usefully presented alongside the national income accounts rather than deducted from GDP. The primary reason for this is that it is unclear to what extent the damages have already been internalised. Hence, if they were subtracted some double counting would result. The second is that by far the greatest benefit of the data is to guide policy on the economic and environmental interface, and the aggregation of damages into GDP is of little value in this regard.

1.2 PROJECT OBJECTIVES

To develop the research further it was recognised that the coverage of existing impacts should be extended as far as possible across the study countries. This is important for policy-making within the EU, where cross-country resource allocation and regulatory decisions have to be taken. Second, *damages need to be attributed to sources*, since this is an essential linkage if the approach is to have real policy relevance. Third, other forms of pollution media need to be investigated such as damages to water and land. Specifically, the main objectives of the work programme which this book reports on were:

(a) To review the credibility of monetary valuation of environmental damage, as well as that of other indicators of environmental impacts and pressure. The review examines the strengths and weaknesses of the different measures, particularly with reference to their role in developing accounts and in their usefulness to decision-makers. In many areas, the methodology has been updated in the light of advances in other areas.

(b) To enhance the valuation of environmental damages in the EU in the light of new data. In GARP I the team found a number of gaps in the data required for the valuation of environmental damages. These missing data varied across countries, with some having better information than others. GARP II fills these gaps as much as possible and obtains valuations that are both credible and comparable to the maximum extent.
(c) To extend the range of pollutants covered by the analysis. In particular, careful consideration was needed of the relationship between primary and secondary pollutants, and the effects of heavy metals on human health was also to be investigated. Impacts on water and land contamination are also evaluated.
(d) To attribute the spatially disaggregated damages to different sources of pollution using a 'multi-source' version of the ECOSENSE model developed by IER, University of Stuttgart. This model allows damages to be assigned to countries and economic activities (sectors), identifying crucial transboundary impacts.
(e) To review critically the main methodological approaches for the estimation of defensive expenditures, together with any estimates that were available. For each stressor (source of damage) there are a number of defensive expenditures that are undertaken. This issue was identified as important in the first phase, but very sparse data were available. Comment is made on how these should be integrated into an accounting system.
(f) To evaluate the replicability of the methodology to other countries in the EU and consider the feasibility of preparing such estimates on a regular basis so as to form an impression of the changing state of the environment over time. In this regard, the final presentation of the data was an important issue with the prime objective being to maintain a high degree of policy relevance.

Progress has been made in all of these areas and it is the purpose of this report to show that the basic methodology has been consolidated and certain limits to the work recognised.

1.3 OVERVIEW OF APPROACH

An overview of the key stages in the work is given in Figure 1.1. As can be seen, the second phase of this project, called GARP II, still maintains a major focus on the impacts of air pollution. Airborne pollutants can be distinguished as either primary or secondary. A primary pollutant is directly emitted from a source (usually a combustion process such as a power

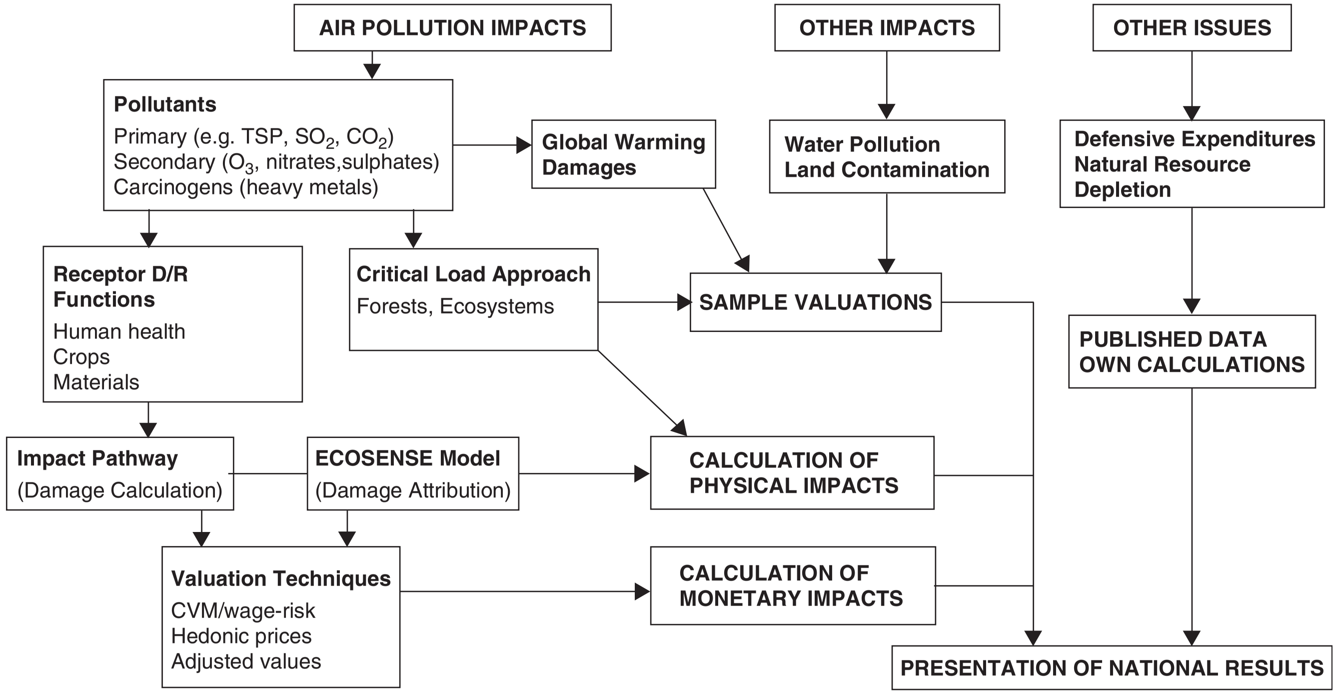

Figure 1.1 Schematic representation of the GARP approach

station, factory or motor vehicle). Examples include sulphur dioxide (SO_2) and total suspended particulates (TSP). Secondary pollutants (such as ozone (O_3), nitrates and sulphates) are formed by chemical reaction in the air. In this analysis, we are concerned with pollution concentrations. Data on *all* these can either be measured (that is, supplied by monitoring stations) or modelled via computer dispersion models.

There are two main elements to the analysis: *damage calculation* and *damage attribution*. Damage calculation involves a restricted impact pathway analysis, whereby concentration data is interpolated to national grid maps and overlaid with population data. This is essentially what was done in GARP I and is in contrast to a full 'bottom-up' (engineering) approach that considers in detail the emissions from site-specific sources. The impact pathway analysis is discussed in more detail in Chapter 3.

The damage attribution exercise, however, does allow emissions to be allocated by economic sector. This is achieved through the ECOSENSE model, an integrated computer tool that incorporates modules for technology, emissions, exposure–response functions and valuation data (see Chapter 8, Section 8.1 for an introduction to the ECOSENSE model). Both damage calculation and damage attribution sets of calculations have been undertaken for impacts on human health, crops and building materials. It is for these receptors that exposure–response functions have been established with most confidence.

The analysis allows both physical and monetary indicators of damage to be generated. Where possible one of a number of valuation techniques can be applied to derive estimates for specific 'endpoints' that are closely related to human welfare. Wherever possible the valuation is based on the theoretical foundation of willingness to pay (WTP) or willingness to accept (WTA). Valuation techniques and issues arising are discussed in more detail in Chapter 5.

Figure 1.1 shows that for forest and ecosystem impacts the complexity of these systems does not permit a comprehensive valuation exercise. Hence the approach discussed above cannot be applied. The broad framework for assessing these impacts was a critical load approach, with nitrogen being the main pollutant considered. The natural environment is classified into eight vegetation classes and used in conjunction with land cover data at a European level. Forest assessment identifies three main types of damage from air pollution: loss of timber, reduced recreational benefits and decreased existence value. All these categories present methodological difficulties, although the situation has improved since Markandya and Pavan (1999). It is now possible to make sample valuations (for example, for specific biotopes) and obtain some implicit values from related studies. These are reported in Chapter 11.

Global warming impacts are also a distinct module. The debate on appropriate quantification of these impacts has advanced considerably in recent years. In Markandya and Pavan (1999) the estimates were made of the present value of future climate change damages in the study countries as a result of global carbon dioxide (CO_2) emissions in 1990. This was undertaken on the basis of published studies. The results have been updated, drawing on more recent literature, especially relating to regional impacts.

In Figure 1.1 'other impacts' assessed include water pollution and land contamination. In the case of water the approach involves linking data sets on river water quality, recreational activities and monetary values that exist for these activities, to provide estimates of recreational damage costs. The analysis has been undertaken, in the first instance, only for the UK. Land contamination issues are likely to be very specific. In accordance with the approach taken in the rest of the project, the ideal approach would be to create 'flow' accounts to assess the change in the value of contaminated land over time. However, initial investigations of the data indicated that this would not be possible. Hence, data on the stock of contaminated land together with illustrative figures on the costs of remediation were collected. For both these types of impacts it has only been possible, at this stage, to make some sample valuations. It is hoped, particularly in the case of water impacts, that the methodology will be expanded to become more comprehensive – expanding the quantification of the impact pathway so that the analysis is more in line with that for air pollution.

Under the heading 'other issues' are concepts that the project team believe to be relevant to the concept of environmental accounting but that were not covered in the core activity of the project GARP II. Defensive expenditures (also referred to as environmental protection expenditures) are understood to be an indicator of the total monetary burden which society bears annually for the regulation of environmental degradation and damages induced by the economic use of the environment in the past and present periods. These expenditures can involve environmental protection and restoration activities, damage avoidance and treatment. Estimates have been made as part of this project and many published studies have been reviewed. Chapter 7 discusses the methodological issues concerning defensive expenditures, and the results obtained, in more detail.

Natural resource depletion is also an important issue and is often viewed as being an important parameter in any debate about 'sustainable' development, an issue which this study does not address directly. This subject has, however, received increased attention in recent years (see, for example, Vaze, 1996) and it is an area that we believe should be explored in subsequent research projects. It is also the subject of a project on Sustainability and the Use of Non-renewable Resources (SAUNER) which combines a theoretical appraisal of sustainable development in the presence of non-renewable resource depletion, with empirical estimates of current resource stocks, and future resource use.[2] The output of SAUNER will include suggested policies to narrow the gap between anticipated resource-use paths and those which are efficient and sustainable. It is anticipated that this may be relevant to the methodological development of resource depletion accountancy.

1.4 METHODOLOGICAL ISSUES

The main purpose of the research reported in this book was to estimate the environmental damages sustained as a result of economic activities and to offer an insight about how the information generated can be used in conjunction with conventional economic accounts, principally estimates of GDP. In this context, it is recognised that there is much controversy as to the suitable definition and interpretation of GDP itself, let alone environmentally adjusted GDP. In view of this and many other uncertainties concerning the estimates themselves, the project team do not advocate subtracting the damage estimates from national income figures. Instead, the team believes that the GARP methodology complements the direction taken in the development of international environmental accounting frameworks and sheds new light on the feasibility of aspects of these frameworks. It is therefore hoped that the future evolution of environmental accounting will benefit from the findings of this project.

The research does take a favourable view on another controversial question, namely whether environmental damages or burdens should be monetised. Our view is that valuation is important since it allows environmental costs to be integrated with traditional costs and benefits using a single denominator. As a result of this work, we hope that the significance of economy–environment linkages will become better understood and that resource allocation decisions will be made on a more efficient basis. The approach to valuation is based on welfare principles as defined in neoclassical microeconomics. For further discussion of underlying principles see European Commission (1999, volumes 1–2).

At the outset we should reiterate that, as for the project GARP I published in Markandya and Pavan (1999), what is being valued is the *total damage* caused by the economic activity. This is done (in most cases) by taking the unit value of the damage and multiplying it by the number of units of the environment lost or damaged. This necessitates defining what the ambient environment would be in the absence of anthropogenic activity, an issue which is discussed further in Chapter 8. In this respect the treatment of environmental services is not different from that of other goods and services, which are valued using market prices times the number of units bought/sold. We should also point out, however, that the unit values reported here *can* also be used for a *marginal valuation*, that is, to look at the damage caused by a small increase in pollution levels. We have focused on total damage because it provides estimates broadly comparable with the components of national income accounts.

The valuation method described above, where we take unit values based on existing prices, is also referred to as the partial equilibrium method of valuation, as it is based on taking existing prices as given and valuing damages by multiplying the changes in quantities by the prices. This has come under some criticism on the grounds that partial-equilibrium assumptions are unlikely to hold at the aggregate level, thus reducing the credibility of the valuation. As a general criticism this is not valid; all measures of national activity such as GDP are based on taking prices as fixed and the welfare basis of such measures has been shown to be sound (Weitzman, 1976). At the same time, there can be some cases where policy makers will want to know whether a major reduction in pollution will bring about a given set of benefits as estimated by the partial equilibrium method. The answer is that there are cases where such an assumption will not be valid and major changes in environmental impacts will cause a substantial change in relative prices and corresponding structural shifts within the economy. Consequently, we have indicated in the report the environmental impacts for which this valuation issue arises in a practical context. In such cases we have also provided some examples of valuation methods that are not based on the partial-equilibrium approach.

As we noted earlier, the monetary valuation in general does not command universal approval. This also applies to the particular form of monetary valuation used in this book – that is, damage estimation based on willingness to pay to avoid the damage or willingness to accept payment as compensation for the damage. Hence it is important to put this work in a broader context and to see what the arguments for and against this approach are. This is done in Chapter 2, where other developments in the area of environmental accounting are reviewed. One project in particular, the 'Green Stamp' project funded by the European Commission – Research Framework Programme, has made an important contribution to this discussion. The focus of this project is an investigation into the feasibility of using avoidance cost data in monetising environmental effects rather than willingness-to-pay measures, favoured in GARP II. Green Stamp also promotes an alternative set of indicators to the satellite national income accounts. These indicators would be in the form of avoidance cost curves, developed though input-output and general equilibrium modelling, which provide estimates of the cost of meeting specified environmental standards.

These goals are referred to as 'sustainability norms'. The models proposed (but not implemented) would then develop cost estimates for various scenarios that move the economy from its present condition to one which respected a specified set of environmentally sustainable standards.

We take the view that the Green Stamp approach is useful to policy makers but it does not obviate the need for damage estimation. First, the costs of meeting specified standards do not tell the policy maker whether those standards should be met. That requires a comparison of the costs of avoidance as well as the environmental damages reduced. Second, the costs of meeting standards that are set by policy makers, *but that are not met*, are only an indication of what society would have to pay to achieve certain environmental goals. If society is not paying this price, it may be suffering more, or less, than the cost. Again, only damage estimation will reveal that. Hence the Green Stamp Approach, while useful, is not by itself a guide to policy. Furthermore its costs are not directly comparable with the GDP estimates of the value of goods and services actually produced.

1.5 SPECIFIC ISSUES FOR GARP II

As stated at the beginning, GARP II had a number of key broad objectives that have been identified. In addition, the project explored a number of specific issues relating to the environmental impacts and environmental accounts and these are contained later in this book. These can be summarised as follows:

(a) The measured concentration data for primary pollutants varies substantially in quality, both between countries and pollutants. This strongly influences the credibility of the interpolation and the resulting confidence in the damage estimates. The underlying chemistry describing the formation of secondary pollutants is also highly complicated. Valuation of impacts is also highly sensitive to background levels of pollution. In this study, a range of background levels has been taken and tests for linearity carried out.

(b) The literature on valuation of damages is still strongly dependent on US studies, although this situation has improved since Markandya and Pavan (1999). Many studies are not transferable to the European context and care has to be taken to ensure that functions and values that are used are realistic to European conditions. The valuation work updates estimates on the basis of some recent European studies.

(c) It was evident from Markandya and Pavan (1999) that impacts on human health dominate the overall impacts of air pollution. The key parameter underlying the valuation for health is the Value of a Statistical Life (VSL), on which there is little agreement. This issue has been debated extensively in recent years. The project team concluded, however, that the related concept of Value of Life Years Lost (VLYL) is more appropriate for the type of impact being experienced. This is principally due to the fact that (a) most mortality cases will be for individuals with already severely reduced life expectancy (see Chapters 8 to 15), and (b) the key effects of chronic mortality are best measured in terms of changes in life years lost. A major effort has been made to consolidate the valuation approach as a whole and efforts have been especially directed to the quantification of chronic mortality impacts.

(d) Detailed reviews of the epidemiological literature revealed considerable doubts over some of the health endpoints adopted in Markandya and Pavan (1999). The causal relationship in many cases proved inconclusive (that is, the pathway could not be fully defined) or it was impossible to avoid double counting. These endpoints have been removed. Subsequently, a number of new endpoints have arisen and results prepared for these.

This book addresses many of the principal weaknesses identified in Markandya and Pavan (1999). We believe that in all of the areas mentioned above significant progress has been made in developing sound methodologies, given the many remaining uncertainties that are inherent in this field of research and policy.

NOTES

1. This earlier project was titled the *G*reen *A*ccounting *R*esearch *P*roject I (GARP I) and was supported financially by the European Commission. The present volume is based on the results obtained during a second phase of this project, financed by the European Commission, titled GARP II, and builds on that work in ways that are elaborated in the chapter.
2. SAUNER is a European Commission financed project undertaken under the direction of Professor Markandya, with colleagues from Germany and Austria.

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2. Developments in green accounting

Anil Markandya, Alistair Hunt and Ian Milborrow

2.1 INTRODUCTION

There is a strong interest, both internationally and within the EU, in the preparation of a coherent set of environmental accounts that establish the impacts of economic and social activities on the environment. The UN System of National Accounts (SNA93) included a set of guidelines for the preparation of a set of 'satellite accounts' to complement the conventional national income accounts. This was followed by a handbook on environmental accounting which set out the methodology to be followed for the valuation of natural resources and environmental degradation caused by anthropogenic activity (United Nations, 1993). This is known as the System for Integrated Environmental and Economic Accounting (SEEA). The emphasis, however, was on the underlying principles to be followed, together with only a few general remarks on various valuation techniques.

Since 1994, the London Group of Environmental and Resource Accounting representing statisticians from OECD-government states and international institutions (World Bank, Eurostat) has met to discuss developments in methodology and practical applications. This group has also been charged by the Statistical Commission of the UN with undertaking a review process for the SEEA, known under the acronym SEEA 2000. This review has already been circulated on the web and is due to be published (United Nations et al. forthcoming).

The European Commission first made its position clear in the paper 'Directions for the EU on Environmental Indicators and Green National Accounting (COM 94, 670)' in which the commitment was made to 'develop Integrated Economic and Environmental Indices of economic performance and environmental pressures of economic sectors within 2–3 years'. The general philosophy of this Communication underlines the need to combine different approaches, such as pressure indicators, physical and monetary accounting, within one overall framework.

The academic and government statistical communities have responded to this direction and much research on the subject of environmental accounts (both theoretical and empirical) has begun in recent years. For the purposes of this project a review of the most prominent methodological guidelines prepared by international institutions and key environmental accounting initiatives at the national level has been undertaken. This review covers:

- Dutch CBS Sustainable National Income (SNI) method and the associated GREENSTAMP Approach
- Dutch CBS National Accounting Matrix including Environmental Accounts (NAMEA)
- UK Environmental Accounts (UKENA)

The SEEA and Système Européen pour le Rassemblement des Informatíons Economiques sur l'Environnement (SERIEE) (European System for the Collection of Economic Information on the Environment) accounting frameworks are considered in more detail in Chapter 7 alongside the discussion on defensive expenditures.

2.2 METHODOLOGICAL DEVELOPMENTS IN THE SNI/GREENSTAMP APPROACHES

GREENSTAMP, an EC DG Research-funded project undertaken by Centraal Bureau voor de Statistiek (CBS) (Dutch Statistical Office), Statistical Office of Germany, C3ED (Université de Versailles), Wuppertal Institute and IOR Germany, has the formal title, 'Methodological Problems in the Calculation of Environmentally Adjusted National Income Figures'. It examines existing approaches to environmental accounting and presents a revised methodology, based largely on the approach developed by Hueting (1989) for the Dutch CBS, to measure Sustainable National Income (SNI).

We have discussed the basic complementarity of GREENSTAMP and GARP in Chapter 1. In order to review the GREENSTAMP work further, we first briefly discuss the concept of Sustainable National Income (SNI). We then go on to outline the approach advocated by Hueting to calculate SNI and subsequent developments made by GREENSTAMP. A number of the methodological problems encountered are discussed, together with the direction in which solutions are sought. Finally, we attempt to draw some tentative conclusions as to the future direction of green accounting.

2.3 THE CONCEPT OF SUSTAINABLE NATIONAL INCOME

The theoretical basis for the correction of National Income Accounts for environmental losses was provided by Weitzman (1976) and Hartwick (1977, 1978). Weitzman showed that net domestic product (consumption plus net investment) is the correct measure of the contribution of current economic activity to welfare over time. Hartwick showed that for the economy to be on a path of sustainable consumption, the value of the depreciation of natural resource stocks must be invested in reproducible capital. Both of these contributions suggest that the value of depletion of natural resource stocks should be deducted from gross income. Two assumptions must then be fulfilled for the resulting figure to represent income correctly in welfare terms. The first is that changes in all capital items affecting future consumption are accounted for, and the second is that consumption is a good measure of welfare.

The demand for 'green accounting' has come about precisely because it is felt that the neglect of the environment in national accounts does, in fact, invalidate these assumptions. Hence, it is argued, net domestic product (or any other related statistic such as national income or GDP) is not a fair welfare measure and, more importantly, by its neglect of the environment it gives the 'wrong signals' to policymakers and the public (Hueting, 1989). Nevertheless, the analysis of Weitzman can guide the development of green accounting, not least because of the distinction between stocks (capital) and flows (consumption), which is of great importance when evaluating environmental change.

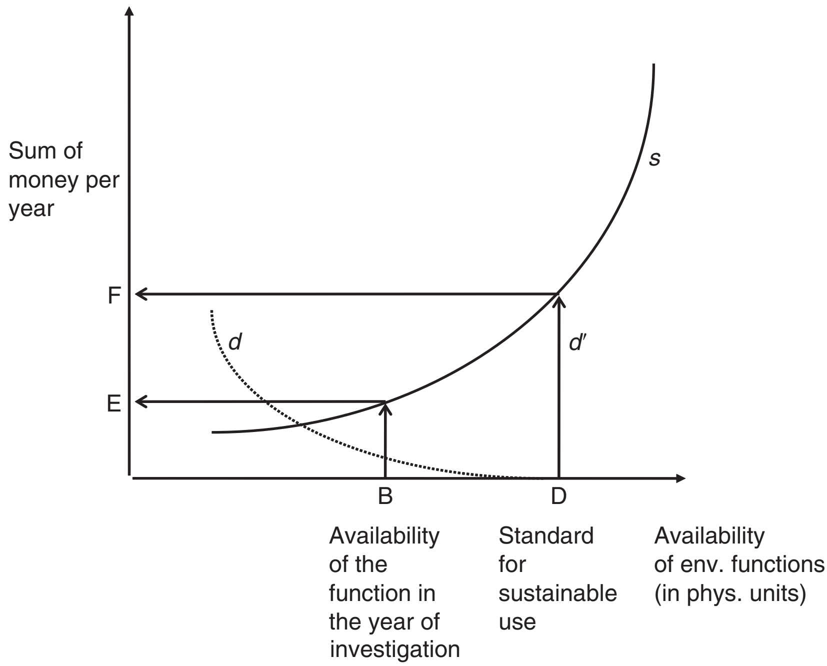
Suggestions for the correction of national income abound. Daly (1989) suggests correcting Net National Product for 'defensive' expenditures and the depletion of natural capital. Daly defines defensive expenditures as 'those regrettable expenditures necessary to protect ourselves from the unwanted side effects of our aggregate production and consumption'. These expenditures should not be counted as final demand, but as intermediate demand. The depletion of natural capital (both non-renewable and renewable) should be treated analogously to the depreciation of man-made capital. Daly is not concerned with the monetary valuation of natural resource depletion, which could be done on the basis of willingness to pay or replacement costs, 'whichever is less'. He argues that the level of arbitrariness in depreciation estimates may not be necessarily larger than those made concerning the depreciation of man-made capital (Daly, 1989, p. 9). Hueting (1989) argues in a similar way, the significant difference being that he advocates the valuation of natural resource depletion (and environmental degradation in general) by using the hypothetical costs of returning the environment to some set of 'sustainable' standards (Hueting, 1989, p. 35).

2.4 THE DUTCH (HUETING) SUSTAINABLE NATIONAL INCOME METHOD AND GREENSTAMP

The main difference between Hueting's Sustainable National Income (SNI) and alternative approaches is the central position of 'sustainability norms' for environmental functions. Hueting reaches back to Hicks's definition of income as the maximum value that a person can consume during a time period and still expect to be as well off at the end of that period as at the beginning (Hicks, 1946). Given environmental degradation and resource depletion, it is the basic premise of the SNI concept that national income can only be sustained if all (or the most important) environmental functions can be sustained forever.[1]

The problem of SNI then becomes one of assessing the maximum level of economic activity that can be developed within an accounting period that respects the sustainability norms. All the costs that are incurred in order to prevent the sustainability norms being exceeded – irrespective of whether they are to be made by industry, government or households – are considered to be intermediate expenditures that therefore do not count as income. To put it simply, SNI is the difference between conventional National Income and the expenditures that need to be made to respect the sustainability norms. Hueting (1989) uses the diagram shown in Figure 2.1 to illustrate his ideas.

The x axis depicts the level of an environmental function, for example, the cleanness of air, the integrity and size of natural habitats, the stock of fish in the sea, all expressed in physical units. The y axis measures money costs or payments per annum. Curve *s* is the supply curve for the environmental function or, equivalently, the prevention cost function. It shows the costs of sustaining a certain level of the environmental function. The social demand curve for the environmental function is *d*. Hueting argues that a complete demand curve for environmental functions (such as *d*), based on individual preferences, cannot be determined. Instead, since many governments, including the Dutch, have adopted 'sustainable development' as official government policy one may then assume, according to Hueting, that society has collectively expressed an absolute preference for the preservation of (certain) environmental functions. Indeed at this moment sustainability norms for the depletion of fossil fuels, climate change, depletion of the ozone layer, acidification, eutrophication, zinc emissions, photochemical ozone, dehydration, and local soil pollution have been proposed by CBS



Source: Hueting et al. (1995b).

Figure 2.1 Demand and supply of environmental functions

and are under review by the Netherlands Institute of Public Health and Environmental Protection (RIVM). However, whilst Hueting believes that the sustainability norms are exogenous, to be determined by science, the GREENSTAMP approach argues that sustainability targets, or 'norms', are constructed through a process of social negotiation and conflict resolution which gives rise to a social demand for sustainability.

This social preference is depicted in the 'collective' demand curve *d*′. Demand for an environmental function is then equal to the sustainability norm, and it is completely inelastic. (Note also that this assumes that environmental sustainability is the binding constraint on environmental degradation – that is, that at D, the marginal utility of the extra unit of consumption that could be generated is less than the marginal utility of the environment.)

Now assume that the present level of the environmental function on the x axis is B. To reach the sustainable level at D, prevention costs of the magnitude of EF have to be incurred. The cost-effectiveness analysis is formulated in terms of abatement cost curves (ACCs). These curves demonstrate for a particular environmental pressure the least cost incurred in reaching its standard by means of available technical environmental measures. In theory, each curve can be plotted by ranking the marginal costs of possible technical measures sequentially according to their increasing magnitude. If this exercise is repeated for all environmental functions that have to be sustained, then the sum of all EFs is the money difference between the conventional National Income and the SNI. This total therefore constitutes an estimate in consumption terms of the opportunity cost of meeting the environmental standards.

The Hueting method estimates an adjusted GDP using existing market prices. These prices do not, therefore, include any allowance for the dynamic effects that would result from a shift in the pattern of economic activity that might be required in moving the economy towards sustainability. The GREENSTAMP approach consequently extends Hueting's work by applying multi-sector dynamic simulation modelling techniques to capture these effects along non-equilibrium paths of economic activity. Thus a key feature of this approach is the setting of different possible future scenarios (in terms of sustainability), and the modelling of the economic paths taken to fulfil them over different time periods.

The GREENSTAMP approach takes the analysis further and argues that the evolution of sustainable development policy cannot easily be determined by the use of a single unit of measure such as the monetary one implicit in the abatement cost, or cost-effectiveness, technique outlined above. It proposes the adoption of a multi-criteria decision support analysis (MCDA) which makes explicit the sort of social choices and ecological and economic trade-offs that might be involved. The cost-effectiveness analysis is therefore complementary to the MCDA rather than the single decision-making measure.

There are certain methodological issues that should be highlighted when considering this approach. They primarily concern the sustainability norm, the prevention cost curve and the general equilibrium consequences of reallocating resources from production to pollution abatement.

2.4.1 The Sustainability Norm

Hueting and GREENSTAMP assert that demand curves for environmental quality based on individual preferences cannot be derived from observed behaviour, since there are no markets for environmental functions, and therefore welfare-theoretic approaches cannot be used. They suggest that, whilst some information on demand for environmental quality can be inferred from defensive expenditure and financial damage, this information is incomplete and often does not address the most vital life-supporting functions of the environment. Furthermore, it is argued that the WTP approach, using non-market valuation techniques, cannot credibly put monetary values on damages that are spread over time and whose significance is sometimes as much ethical as biophysical. More particularly, global problems are not thought to be amenable to valuation based on willingness-to-pay principles.

Hueting/GREENSTAMP note that at the United Nations Conference on Environment and Development (UNCED) Earth Summit in 1992 governments from all over the world adopted the principle of sustainable development. These governments have promised to try and encourage a sustainable use of the environment and of natural resources. Although they do not always seem to act accordingly, Hueting assumes that governments take environmental sustainability seriously and that it [sustainability] is therefore a true reflection of collective preferences for the quality of the environment. The alternative demand function suggested is therefore expressed in the form of sustainability norms for each environmental service, subject to sustainability constraints in its use.

We recognise that it is not possible to apply the welfare-theoretic valuation approach to all types of environmental impacts and that in such instances alternative approaches are necessary. However, it is not obvious that the GREENSTAMP approach is significantly different in its outcome from the welfare-theoretic approach. Both take the policy goal as being a function of a collective social demand derived from some expression of individual preferences. The welfare-theoretic approach makes explicit the individual preferences in monetary terms, whilst the GREENSTAMP approach expresses the (negotiated) demand in physical terms that inevitably have an implicit monetary cost (to their achievement). Furthermore, there are reasons why the GREENSTAMP approach may be less reliable in forming an aggregate social demand function. In particular, a reliance on negotiated demand lays the approach open to influence by relatively strong interest groups and risks resulting in a demand function that misrepresents the interests of individuals within society as a whole.

The welfare-theoretic approach provides a direct measure of monetary worth and therefore allows the importance of different types of environmental damage to be assessed on a comparable basis. It must be recognised that the restrictions that need to be placed on the formulation of the neoclassical growth-with-natural-capital model and the inter-temporal dimensions of sustainability (as the GREENSTAMP approach would require) make it difficult, or impossible, to adopt these monetary measures as sustainability indicators. The GARP project does not have these ambitions. Rather, it seeks to report changes in human welfare through estimation of damage costs on a (willingness-to-pay) welfare basis. We do not advocate integrating the estimates to produce a measure of *green* GDP since it is clear that the techniques have not advanced sufficiently to undertake this work and obvious questions arise as to whether this is methodologically defensible.

2.4.2 The Prevention Cost Curve and the Cost-Effectiveness Approach

As outlined above, the prevention – or avoidance – cost curve maps out the relationship between the level of an environmental function and the social costs that are needed to restore or maintain this level. In the Hueting/GREENSTAMP methodology, costs can accrue from two different sets of actions: technical measures (for example, 'end of pipe' measures, process changes or the development of alternatives for non-renewable resources), and volume reductions in the polluting or extracting economic activities themselves. These would be carried out when technical measures alone are not sufficient to reach the sustainability norm or when they are more expensive than the volume adjustments. Work is currently being done (for example, Meyer and Ewerhart, 1996; CBS, 1997a; Verbruggen et al., 1997) to develop computable general equilibrium models to help this quantification.

An initial study of the theoretical basis and empirical findings of the cost-effectiveness approach suggests that there are significant difficulties to be overcome before the approach can be widely adopted.

The methodological approach required to compute empirical estimates of the ACCs raises certain issues. The availability of statistics on costs that can be aggregated from the firm (micro) level to the branch (meso) level required in the compilation of national income statistics has proved in practice to be very limited. Indeed the GREENSTAMP project included an empirical application of the approach for various compounds of nitrogen, using German data which found that there were in fact 'no usable cost statistics available in Germany'. It reports that 'a lot of intermediate steps related to many uncertainties and assumptions are necessary to come to a common classification level of both the physical and economic side'. It is likely that these difficulties will be greatly exacerbated when, in bringing about volume reductions, an entirely new resource mix exists in the economy. Clearly, in order to successfully model future GDP scenarios the cost parameters are of crucial importance, yet these are highly uncertain at the present time.

It should also be noted that costs are determined in a market through the expression of willingness to pay of purchasers. Thus the costs of pollution abatement technologies will still reflect individual preferences for environmental protection as far as they are realised. (However, only if these costs are actually incurred can they be interpreted as a valuation in any sense since, if they are not, there is no expression of preference for the technologies made and therefore no opportunity cost identified.) Indeed, in an (admittedly theoretical) perfectly competitive world it is a standard microeconomic conclusion that prices equate consumer preferences with opportunity costs at the margin. Whilst we do not pretend that this solution holds in the real world, beset by market imperfections as it is, we think that more attention should be given to definition of the practical meaning of the opportunity cost concept before avoidance cost techniques are widely adopted.

2.4.3 Conclusions on the Hueting/GREENSTAMP Approaches

The two broad approaches taken to develop monetary indicators for the European environment, exemplified by the GARP and GREENSTAMP, have different end objectives. GARP investigates the scope for adopting satellite accounts, which draw out the reported linkages between economic sectoral activity and environmental damage, whereas GREENSTAMP models cost-effective paths from patterns of current economic activity to ones which allow sustainability goals to be met. The GREENSTAMP approach is therefore more ambitious in its intentions for policy guidance, having the future achievement of sustainable development as its explicit objective. GARP, in highlighting the contribution that individual sources make to the damage cost, implicitly points to those areas of economic activities that should attract environmental policy initiatives but does not seek to define levels of damage in terms of sustainability. From the theoretical and empirical developments that have been reported to date, we suggest that there are a number of considerations to be made when defining future research needs in this area.

First, it is clear that any approach to monetising environmental impacts for policy indicator purposes will be subject to problems of both data availability and validity. We do not think that the logistical difficulties this creates are markedly different in the two approaches. Indeed, we suspect that data scarcity is most extreme for the same environmental impacts in both approaches. It is clear, for example, that global issues (for example, global warming and biodiversity) present severe difficulties for achieving credible monetary valuation.

If this first point is accepted, then it follows that the usefulness of the different policy goals has to be assessed. We would argue that at present the concept of sustainable development remains too ill-defined for monetary values to play a useful part in its measurement. It would therefore be preferable to maintain a two-tier research programme consisting of (a) the development of monetary valuation methodologies and some agreement as to their useful scope across the range of environmental indicators, and (b) further consolidation of the type of work currently being undertaken by Eurostat on environmental pressure indices particularly in their application as indicators of sustainable development.

We suggest that the monetisation of environmental impacts for green accounting can effectively be related to (and most easily understood in) the policy decision-making process when expressed in welfare terms. It is believed that welfare-based values are generally appropriate in political negotiation procedures as well as having a sound theoretical basis. This, combined with the recognition of data constraints made above, suggests that, where possible, non-marketed, as well as marketed, impacts should be valued directly. It is also recognised, however, that estimates of the economic avoidance costs incurred in order to achieve environmental objectives are essential in informing policy decisions.

2.5 SHOULD DEFENSIVE EXPENDITURES BE SUBTRACTED FROM NATIONAL INCOME?

A key question regarding the Hueting approach is whether it is correct to subtract defensive expenditures or pollution prevention costs from National Income, to arrive at a measure of Sustainable National Income.[2]

In a seminal paper, Mäler (1991) has considered the subject of green accounting from a welfare-theoretical perspective. Mäler discusses corrections on Net National Product (NNP). Apart from his treatment of labour, which is not relevant for the present discussion, his four main conclusions are that:

1. the value of environmental services to households and the direct use of natural resources by households should be included in NNP, valued at households' marginal valuation. If the value of environmental services is thus included in NNP, defensive expenditures must not be deducted – to avoid double-counting;
2. the value of environmental services and natural resources to firms is already accounted for in NNP through the value of the output of firms;
3. the value of input goods to increase environmental quality should be deducted from NNP; they are intermediary goods (note that input goods that are used by firms for this purpose are already treated as intermediary in conventional NNP);
4. the value of the change in environmental and natural resource stocks should be included in NNP, valued at its (discounted) future value to households and production.

Mäler's conclusions emphasise the difference for accounting purposes of environmental services to households and to production. This is clear after a moment's reflection. The value of environmental services to production, and also its inverse, the environmental damage to production, is already accounted for in the value of production. If due to air pollution (for example, ozone) the value of crop production is less than would have been the case without pollution, then NNP (or GDP) is also less than would have been the case without pollution. Deduction of this damage from NNP would thus be double-counting.[3]

Defensive expenditures must *not* be deducted from NNP if the value of environmental services is included in NNP. Given the state of the environment, defensive expenditures must increase welfare (otherwise they would not have been undertaken). In conventional NNP defensive expenditures are credited without environmental damages being debited. If, however, environmental damage *is* debited (as proposed by Mäler), then it would be technically wrong not to credit defensive expenditures (that is, include them in GDP).

An important entry in Mäler's accounts is the volume of environmental services, valued at households' marginal valuation, though he does not elaborate on this entry. Although it might be possible to measure willingness to pay for *changes* in environmental services, it seems rather difficult to assess the value of total environmental services. In an attempt to find a way out, Hamilton (1996) and Atkinson (1995) suggest that where households are able to make defensive expenditures in response to environmental degradation, the value of these expenditures may be an indicator of the value of environmental services (Atkinson, 1995, p. 5). These defensive expenditures by households are already accounted for in NNP: they are included in household expenditure. The correction that Atkinson and Hamilton propose comes down to deducting net pollution,[4] valued at households' marginal valuation, from NNP. This solution, although practical, rests of course on the heroic assumption that defensive expenditures are a proxy for the marginal value of environmental services.

The change in environmental and natural resource *stocks* must be valued at their future (discounted) value to households and production. In a theoretical model with perfect foresight, such as Weitzman's, this poses no special problems. In the real world however, the future is uncertain. To use only one example, it is quite unclear how a loss of biodiversity today will affect future production possibilities and household utility. Because of a fundamental lack of knowledge of the future, both at the ecological and economic levels, the value of stocks can only be speculated upon. In the sustainability debate, many have therefore proposed some form of 'safe minimum standards' for the depletion of environmental and natural resource stocks. Safe minimum standards and marginal valuation cannot be reconciled. It is difficult to imagine how safe minimum standards could be used in the accounting framework of Mäler.

In the present situation, where environmental services are not included in GDP, the treatment of input goods is at best inconsistent and the value of changes in environmental stocks is rarely accounted for, it is best, in our opinion, to report on defensive expenditures without making a formal adjustment to GDP. This position is consistent with our view of how to treat damages to households from reduced environmental services, which was set out in Chapter 1. Defensive expenditures are discussed in more detail in Chapter 7.

2.6 NAMEA SYSTEM

Another approach to environmental accounting is the presentation of environmental stocks and flows in non-monetary units in so-called 'satellite accounts'. The Statistical Bureau of the Netherlands has developed an accounting framework, the National Accounting Matrix including Environmental Accounts (NAMEA), in which monetary information on the economy and *physical* information on the environment have been partially integrated (Keuning, 1993).

NAMEA combines the economic accounts of the National Accounting Matrix (NAM) with environmental indicators. Table 2.1 presents the main structure of NAMEA in schematic form. The *economic* accounts of NAMEA are: the goods and services account (1), consumption account (2), production account (3), income generation account (4), distribution and use account (5), capital account (6), financial balances (7), tax account (8), and rest of the world (ROW) accounts (9 and 10). For each account the receipts are presented in rows and outlays are presented in columns. Each account balances total receipts and total outlays. Important balancing items are: net domestic product (4, 3), net generated income (5, 4), net savings (6, 5), surplus or deficit on the current account of the balance of payments (9, 10).

NAMEA contains two *environmental* accounts: an account of substances (11), and an account of environmental themes (12). These accounts are expressed in physical units, therefore they do not influence the monetary row and column totals of the economic accounts (1–10). The row sums of the environmental accounts correspond with the totals in the columns. The substances account (11) contains 13 substances: CO_2, N_2O, CH_4, CFCs and halons, NO_x, SO_2, NH_3, P, N, solid waste, waste water, natural gas and oil. The columns reflect the origin of the substances, the rows reflect their

Table 2.1 Main structure of NAMEA

	Goods and services 1	Cons. 2	Production 3	Capital 6	ROW current account 9	Substances 11	Themes 12	Total
1								
2						Emissions by consumers		
3						Emissions by producers		
6						Other emissions and increase of natural resources		
9						Emissions from ROW		
11			Absorption of substances		Emissions to ROW		Contri-bution of substances to themes	Destination of substances
12				Theme-equivalents				Theme-equivalents
Total						Origin of substances	Theme-equivalents	

destination. Polluting substances are generated by households (2, 11), firms (3, 11) and other sources (6, 11). Stocks of natural resources (natural gas, oil) can change (6, 11). Transboundary pollution is presented in the ROW accounts; imports of pollutants (9, 11) and exports of pollutants (11, 9). Finally, a number of emitted polluting substances is absorbed in production processes, for example, the purification of waste water or the incineration of wastes (11, 3).[5]

The themes account (12) presents indicators of the following environmental themes: the greenhouse effect, depletion of the ozone layer, acidification, eutrophication, the accumulation of waste, waste water, and the depletion of natural resources, that is, fossil fuels. The substances of account (11) are weighted with theme-related environmental stress equivalents and then aggregated column-wise by theme (11, 12). The environmental theme indicators (in theme-equivalents) are presented in the column of the capital account (12, 6).

NAMEAs for the Netherlands are annually published in the National Accounts. The NAMEA of 1994 is published in the National Accounts of 1996 (CBS, 1997b). Such information is useful but, lacking monetary values for environmental variables, it limits the extent to which economy and environment can be integrated.

2.7 UK ENVIRONMENTAL ACCOUNTS (UKENA)

The Office for National Statistics (ONS) established an environmental accounting unit in July 1995 and published the first phase of work in the following year (Vaze and Balchin, 1996). This overview is a summary of that work. The basic aim is to explore economy–environment interactions with the long-term aim of preparing systematic and comprehensive accounts that illustrate the pressures placed on the environment by economic activities. The pilot set of accounts contained data on atmospheric emissions disaggregated by industries, stocks of natural resources and some details on environmental protection expenditures.

In addition, the Department of the Environment (DoE, 1996) published a set of sustainable development indicators designed to highlight some of the key trends in the state of the environment and in stocks of natural resources (state indicators); environment–economy interactions (pressure indicators); and the measures which are being taken to mitigate adverse impacts, including expenditure on pollution abatement (response indicators). These indicators are intended to reflect current policy concerns and are seen by ONS as being complementary to the UKENA.

This section considers the first topic of attributing airborne emissions to industrial sectors since this is most in line with the emphasis of the GARP approach. The project team considered the paper on valuation of oil and gas resources (Vaze, 1996) to be a valuable contribution but did not feel that this was an area for further analysis. Environmental protection expenditures fall under the theme of defensive expenditures which is considered separately in Chapters 7 and 15.

2.7.1 Structure of UKENA

The pilot environmental account follows quite closely the Dutch NAMEA system as presented in the previous section. UKENA is based around an input–output representation of the economy based on the Standard Industrial Classification (SIC92) which is consistent with NACE Revision 1 used by Eurostat. The structure is an *external satellite account* that considers activities outside the production boundary, as delimited by the standard national economic accounts.[6] The UKENA is *external* in the sense that activities inside and outside the production boundary are linked, primarily through the operations of industry. An illustration is provided in Figure 2.2. Some components are contained within the production boundary (for example, defensive expenditures) and others are outside (for example, airborne emissions and natural resource valuations).

ONS do not take the view that environmental protection expenditures should be deducted from GDP (even though some of these could be regarded as 'defensive') since this expenditure still creates multiplier flows and economic benefits within the economy. Airborne emissions data are presented in physical units. There is an interest, however, in considering monetary valuation as an extension to the work once the debates over how best to undertake it are resolved. In addition, ONS identify issues for the future to be the assessment of incremental costs due to environmental degradation and pollution abatement expenditures.

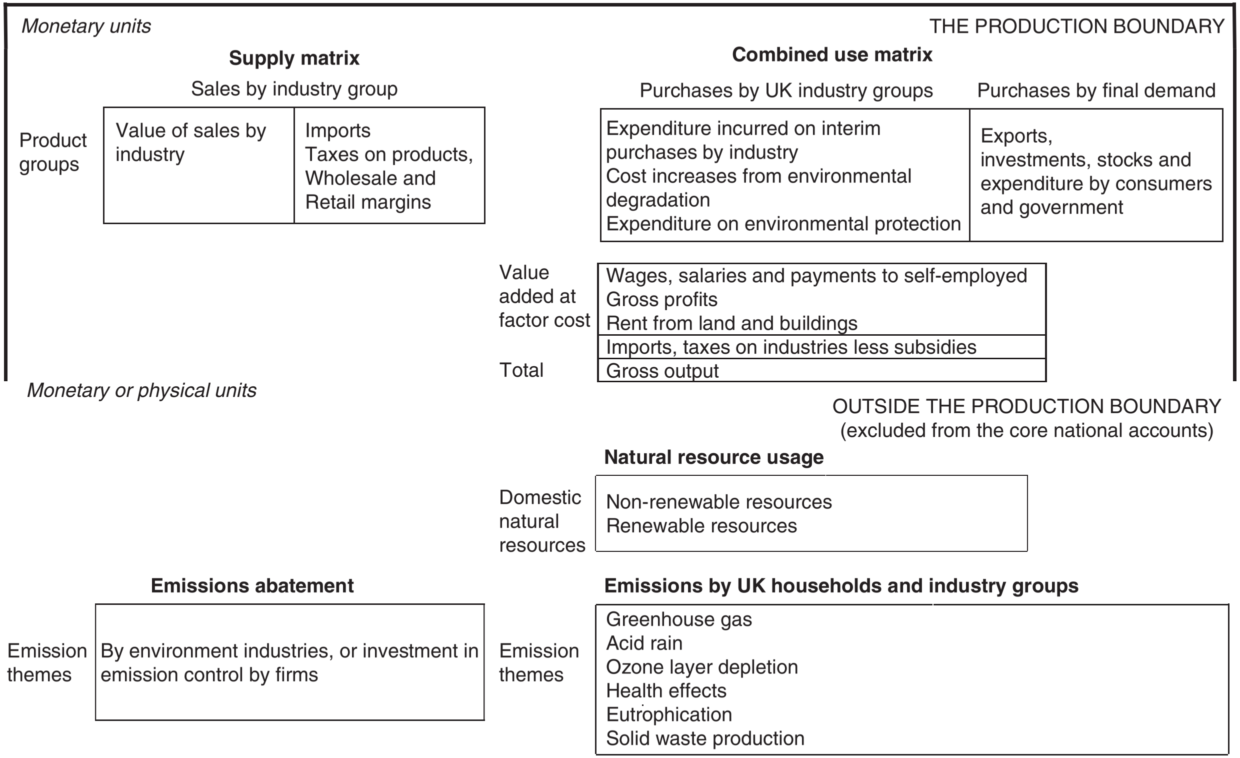
Atmospheric emissions are reported according to their contribution to a number of *environmental themes*. The objective of this approach is to link data on economic activity to widely understood environmental issues, particularly those that are prominent in the minds of policymakers. Emissions have been weighted on the basis of scientific advice, yet the relative contribution of each pollutant to total environmental damage is recognised as being very uncertain. The themes used so far are discussed below.

Global Warming The greenhouse gases CO_2, CH_4 and N_2O were aggregated after being weighted according to 100-year global warming potentials relative to the CO_2 reference. These weights are taken from IPCC (1995), which reports the uncertainty of the values to be around ± 35 per cent.

Acid Rain Precursors This theme was created by weighting SO_x, NO_x and ammonia on the basis of molecular weight and assuming that the gases would dissociate causing acidity in the ratio 2:1:1 and using SO_2 as the reference.

Ozone Depletion The weighting of CFCs and HCFCs and other halogenated compounds assumed that all users were using an average mix of the different fluids. CFC-11 (although technically phased out in the UK since January 1994) was used as the reference. These data are presented separately since they are not consistent with the standard reporting year used in the UKENA (1994).

In addition, data are recorded by specific airborne pollutants: particulates (black smoke), CO, benzene, lead and volatile organic compounds (VOCs). The actual method of calculation is quite simple and involves looking at



Source: Vaze and Balchin (1996).

Figure 2.2 Links between the environmental accounts and the national accounts standard economic and energy consumption data in conjunction with published emissions coefficients (that is, few emissions are monitored at source). The approach necessitates a simplification of pollution chemistry and the responses of different receptors cannot be distinguished. Hence, it is not possible to derive damage estimates.

The UKENA is a major initiative in the formulation of a set of satellite environmental accounts at the national level. The underlying assumption is that GDP is a measure of production, not welfare, hence the focus is more on the *emitters* of pollution (economic activities) as opposed to *receptors* of pollution (categories of damage). The UKENA is accompanied by input–output matrices at a high level of disaggregation. This facilitates integrated economy–environment modelling such as the possibility of attributing the emissions of the electricity generation sector to its customers. In addition, since internationally agreed nomenclature is used, environmental pressures exerted by an industry can be easily compared with economic data sets such as employment, investment or trade. International comparisons are also possible.

2.8 CONCLUSIONS

The major developments in green accounting have been with respect to (a) measures of sustainable national income based on sustainability standards, (b) systems of physical accounts for environmental stocks and flows as they relate to the economic activities and (c) measurement of expenditures on environmental protection and pollution abatement.

GARP II complements this work by focusing on monetary damage estimation. It may be possible in the future to link the GARP work to UKENA. The UKENA method of focusing on a number of well-recognised themes is sensible and indeed attractive, even if in doing so the degree of detail is compromised. In comparison, GARP offers a high level of technical detail and is based on firm technical and economic foundations. A valid compromise is needed, therefore, between policy relevance, theoretical consistency and statistical reliability. The relevance of GARP to UK environmental statistics was recognised when the Office of National Statistics published a summary of GARP I results in its review of UK Environmental Accounts (Markandya and Milborrow, 1998).

It might be possible to use the receptors considered in GARP and present a separate set of results at a somewhat semi-aggregated level, along the lines of the *themes* approach taken by UKENA. It would probably be easiest to do this for acid rain precursors. This would incorporate the receptors of materials, crops, forests and ecosystems.

It is the intention of the UK ONS to calculate emissions accounts for previous years and to investigate a number of new themes such as inland water systems, coastal/marine systems and waste.

NOTES

1. The concept of Sustainable National Income turns out to be more complex than was first envisaged. For a recent discussion of alternative Hicksian-type measures see Asheim (1998) and Pezzey (1998).
2. The term defensive expenditure is used concurrently with environmental control expenditure. The precise meaning in each case has to be ascertained from the context in which it is being applied.
3. In our approach the reduction in production due to pollution is measured and reported alongside the GDP figures to indicate how much pollution affects productive potential.
4. Net pollution is defined as emissions of pollutants generated by production and consumption less the dissipation of pollutants in the environment in a given time period (Atkinson, 1995, p. 3).
5. The emissions from activities such as waste water purification and waste incineration are presented in the production account (3, 11).
6. This is in agreement with the SNA93 which recommends the preparation of satellite accounts as opposed to simply extending the conventional boundaries.

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3. Developments in pathway analysis

Paul Watkiss, Mike Holland and Katie King

3.1 INTRODUCTION

There have been a number of developments in the impact pathway analysis, pioneered by the ExternE and Green Accounting research projects, since the last phase of GARP was completed. Some relate to methodological issues, such as a significant increase in the number of effects that are quantifiable, or the description of uncertainty. Others relate not to strict methodological developments, but (for example) to a growing acceptance of the methodology as being sufficiently robust to be useful for informing policy. This chapter highlights the following major developments:

- increased acceptance of the methodology;
- increase in the range of pollutants and effects that can be considered;
- updating exposure–response functions;
- treatment of uncertainty;
- improved modelling and reporting of air quality;
- attribution of damages to economic sectors;
- quantification of year on year changes in damages.

3.2 ACCEPTANCE OF THE METHODOLOGY: APPLICATION IN POLICY ANALYSIS

The methodology used in GARP is closely linked to that of the European Commission ExternE Project. The methodological outputs of the two studies, taken together, have now been used in a variety of policy applications, covering environmental quality objectives, emission limits and the energy sector. These include:

- economic evaluation of a draft directive on waste incineration (AEA Technology, IER, ARMINES, Electrowatt, 1996);
- economic evaluation of ambient air quality limits for SO_2, NO_2, fine particles and lead (Instituut voor Milieuvraagstukken (IVM)

(Institute for Environmental Studies), Norsk Institutt for Luftforskning (NILU) (Norwegian Institute for Air Research) and International Institute for Applied Systems Analysis (IIASA) 1997);
- external costs and electricity taxation (VITO, see CIEMAT, 1998);
- introduction of externalities into the electricity dispatch system in Spain (CIEMAT and IIT, see CIEMAT, 1998);
- incinerators and cars: a comparison of emissions and damages (ARMINES, see CIEMAT, 1998);
- social costing and the competitiveness of renewable energies for remote autonomous communities (NTUA, see CIEMAT, 1998);
- solid waste incineration versus landfilling (IEFE, see CIEMAT, 1998);
- externalities of energy scenarios in the Netherlands (IVM, see CIEMAT, 1998);
- cost–benefit analysis of measures to reduce air pollution and decision on building gas-fired power plants in Norway (ENCO, see CIEMAT, 1998);
- strategies for meeting future electricity demand in São Miguel Island (Azores archipelago) (CEEETA, see CIEMAT, 1998);
- cost–benefit analysis of the UNECE Multi-pollutant, Multi-effect Protocol (AEA Technology, Eyre Energy Environment and Metroeconomica, 1998);
- comparing costs and environmental benefits of strategies to combat acidification in Europe (Krewitt et al., 1998);
- economic evaluation of acidification and ground level ozone (AEA Technology, 1998);
- development of a simplified methodology for assessment of externalities in developing countries (Spadaro and Markandya, 1998).

In addition the methodology has now been applied in a number of countries outside the EU, including the US, Canada and a growing number of developing countries. The list is not exhaustive, but it does demonstrate that in some areas the work has gone beyond being just a research activity. Furthermore related activities in the US have resulted in similar methodologies being applied there.[1] Increased use of the impact pathway approach has opened the methodology up to much wider review than previously, some of which has, inevitably, been critical. Investigation of much of this criticism has shown it to relate to misunderstanding of the methods used. Some criticism has also related to concerns over the presentation of results for effects where there remains significant uncertainty. A variety of approaches have therefore been investigated for presenting data in ways that convey uncertainties more transparently.

3.3 RANGE OF EFFECTS NOW QUANTIFIABLE

Earlier work in Markandya and Pavan (1999) established that it was possible to quantify the following environmental costs:

- effects of fine particles on health and buildings (through soiling);
- effects of SO_2 on health, materials and crops. Effects on forests were also quantified, but results were reported as being too uncertain for application elsewhere;
- effects of ozone on health and crops;
- fertilisation effects of nitrogen (N) deposition on agriculture;
- global warming impacts of greenhouse gas emissions;
- reduced amenity from exposure to noise.

The impact pathway analysis used in Markandya and Pavan (1999) has now been extended to cover other pollutants, such as the effects of carcinogens on health (including benzene and heavy metals). In addition, there has been considerable work on the development of methods to assess:

- water quality; and
- contaminated land.

Additional work has also been undertaken on defensive expenditures, on valuation of forests and ecosystems and on global warming (with the analysis of country-specific damages).

3.4 UPDATING EXPOSURE–RESPONSE FUNCTIONS AND VALUATION DATA

Exposure–response functions and valuation data used in the earlier report have been kept under constant review to ensure that those used reflect the latest state of the art in impact assessment and valuation. Without this the output of the study would rapidly become outdated and of little use because of developments elsewhere. A major asset of the present study is the fact that it acts as a resource for a large amount of data. The scope of the database is unrivalled, partly because of the broad scope of the study, and partly because of the open approach taken to reporting uncertainties. Other data sets of dose–response functions have tended to focus on single issues (particular pollutants and particular receptors). They have also tended to take a restrictive view on uncertainty, restricting function selection to effects that have become widely accepted. In some ways this could be said to represent good science, in accepting that traditional standards for scientific proof should hold. In other ways, however, this approach is questionable, as it ignores the output of some more isolated, but well-conducted studies. Here, we have preferred to take the option of quantifying damages where quantification based on results of well-conducted studies is possible, and reporting uncertainties in such a way that the reader should gain a good sense of the reliability of the estimates made. We take the perspective that a failure to quantify effects implicitly provides a zero valuation and that where this conflicts with available information it is better to provide estimates with supporting information on likely errors.

3.5 TREATMENT OF UNCERTAINTY

Whilst our earlier report discussed the uncertainties of the impact pathway analysis in some detail, the methodology at that time did not permit these component uncertainties to be integrated in such a way as to provide a properly quantified guide to the reliability of the results for each impact.

Significant progress was made in the ExternE Project (see Rabl, in European Commission, 1998, chapter 5). Quantified confidence bands are now available for each effect considered, based on the use of geometric standard deviations and confidence intervals based on the log-normal distribution.

The above method would be ideal if it could be applied consistently for the many impacts that are reported in this book. In fact the data required are frequently missing and much of the uncertainty cannot be integrated precisely or transparently within such a statistical framework. This typically concerns discrete choices on issues such as the approach adopted for valuation of mortality, or discount rate. These issues are best dealt with by providing qualitative uncertainty assessments and, where appropriate, using explicit sensitivity analysis. Both these responses to uncertainty have been taken in this report. In Chapter 4 and elsewhere the reliability of specific dose–response functions or other elements of the valuation chain are given rankings on a scale of 'A' to 'C'. Discussions led by Ari Rabl within ExternE led to these categories being related to the geometric standard deviations σ_G and confidence intervals described above. The labels are:

A = high confidence, roughly corresponding to $\sigma_G = 2.5$ to 4;
B = medium confidence, roughly corresponding to $\sigma_G = 4$ to 6;
C = low confidence, roughly corresponding to $\sigma_G = 6$ to 12.

The main results of the sensitivity runs are reported in Chapter 8, where damages from pollutants for which the dose–response functions are not well determined are examined, along with alternative exposure functions for pollutants which are generally accepted as having damaging impacts.

3.6 IMPROVED MODELLING AND REPORTING OF AIR QUALITY

An increase in activity relating to air quality mapping (promoted through increased regulation in this area) has enabled a number of improvements in this report. For example, the UK now has coverage of more pollutants than before. Although pollution maps for Italy are not as complete, the availability of data has improved significantly compared to the situation that existed in 1994 when the earlier estimates were made.

In the near future the results of studies to compare the different measurement methods used to monitor air quality in different countries around Europe should become available. It is known that there is variability in measurement in Europe, limiting the extent to which comparison of results can be made between countries. This affects measurement of PM_{10} (particulate matter with a diameter of 10 micrograms or less) in particular, because of the manner in which particles are collected and conditioned in monitors.

3.7 ATTRIBUTION OF DAMAGES TO ECONOMIC SECTORS

Attribution of damage to the environmental sector responsible for causing it was an obvious next step from GARP I, when damages were reported only on a recipient basis. Some attempts had been made to do this before, but these relied on extrapolation of damages on an ECU or dollars per tonne of pollutant basis. That type of approach is flawed because it fails to account for the site specificity of impacts relative to the location at which the pollutant (or other burden) entered the environment. To do this properly it is therefore necessary for modelling work to retain site specificity within the analysis. This has now been made possible through developments in the EcoSense software developed at IER. Similar software has also been developed in the US (the USEPA has something similar, for example, developed by Abt Associates).

That this is important can be shown by reference to results of the ExternE National Implementation Programme, where extensive variations between plants within and between countries was found (CIEMAT, 1998). This variation, combined with the fact that, within any country, plant from particular industrial sectors are often clustered together, shows that it would be wrong to assume either that an individual plant is 'representative', or that errors will average out.

3.8 QUANTIFICATION OF YEAR ON YEAR CHANGES IN DAMAGES

Again, this is an obvious next step following GARP I. Implementing it, however, has been surprisingly difficult in many cases because of delays in production of emission inventories, and variation in the methods used for developing the inventories from year to year. The importance of this last factor is obvious from the expectation that year on year change will be of the order of only a few per cent, even when (as now) there are major actions under way to reduce emissions. Variation in methodology for any year can easily lead to changes in excess of a few per cent, removing the validity of a year on year comparison. In this report, fortunately, we have been successful in providing results from a number of years for the UK, as demonstration of the results that can be obtained.

NOTE

1. Note that even in these areas research will continue to be needed in order that the methodology is kept up to date with the latest research findings.

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4. Updates to exposure–response functions

Fintan Hurley, David Howard, Paul Watkiss and Mike Holland

This chapter reports the exposure–response functions used in estimating the loss of environmental services resulting from economic activity in each country. These functions include some major revisions since the first phase of this project, particularly in the treatment of chronic effects of particle exposure on mortality. All the exposure–response functions used in this study were taken from the ExternE Maintenance and Transport projects, which in turn have built on the earlier ExternE work that was the basis of GARP I. The functions used in the main analysis are presented below by receptor category.

4.1 HEALTH

Since the analysis in GARP I, there has been significant progress on the quantification and valuation of all health effects. On quantification, there has been a major revision of exposure–response functions for the classical pollutants: particulate matter (PM), ozone (O_3), sulphur dioxide (SO_2), oxides of nitrogen (NOx) and carbon monoxide. It is still the case that the main evidence refers to the more-or-less immediate effects of daily levels of these pollutants (*acute effects*). There has however been substantial progress in understanding how best to use the available evidence regarding how long-term exposure to PM may affect mortality (*chronic mortality*): this is considered separately below.

As stated above, the updated assessments of the emerging epidemiological evidence in this book draw heavily on work carried out for the ExternE programme. Early ExternE work was heavily indebted to quantification by the US epidemiologist Bart Ostro, and relied on US studies which dominated the literature at that time, referred to by the European Commission DGXII (1995). The updated ExternE functions draw where practicable on European studies, and especially on the major EC-sponsored APHEA research programme (Katsouyanni et al., 1995).

The background to and selection of exposure–response (E-R) functions is described briefly in the following sections. They cover functions used in the core implementations of GARP II (see Table 4.1) and in the sensitivity analyses (see Section 8.4). Parts of the present summary are based on ExternE-related surveys, in particular Donnan and Hurley (1997) for ExternE Maintenance, and Pilkington et al. (1997) for ExternE Transport. Finally, we consider briefly the effects of long-term exposures to other pollutants, especially metals where cancer is the main endpoint evaluated.

4.2 METHODOLOGICAL ISSUES: THE CLASSICAL POLLUTANTS

4.2.1 Strategy

We have surveyed the epidemiological literature to assess what pollutants have been shown to be associated with what adverse health effects, and in particular which associations are quantifiable. This involves choosing exposure–response (E-R) functions from specific well-conducted studies, and where appropriate adapting them where evidence suggests this may improve transferability. Functions were selected before the APHEA meta-analysis results were available, and we did not carry out formal meta-analyses separately. Rather, we selected E-R functions from individual studies to give numerical estimates of physical damages at or near the average of available good studies. Table 4.1 summarises the E-Rs, and includes a judgement of the embodied uncertainty on a scale of A to C.

It is well-known that association does not necessarily imply causality; a real relationship with one pollutant might show as an association with another. We wanted to avoid the double-counting that would arise if the same health effect was wrongly attributed to more than one pollutant, and the effects then added. Consequently some statistically significant E-R relationships, especially for NOx and CO, have been included only in sensitivity analyses. On the other hand we have included some E-R functions (for example, linking PM or O_3 with restricted activity days (RADs), or PM with chronic bronchitis) where the direct evidence is weak, but a relationship is to be expected in the light of strong evidence for related endpoints. This is an application of the *coherence* argument of Bates (1992).

Judgements such as these are a focus of debate currently among scientists and policy makers concerned with the health effects of air pollution. The following notes highlight some of the issues.

Table 4.1 Quantification of human health impacts

Receptor	Impact category	Reference	Pollutant	f_{er} [1]	Uncertainty rating
ASTHMATICS (3.5% of population)					
adults	Bronchodilator usage	Dusseldorp et al., 1995	PM_{10}	0.163	B
	Cough	Dusseldorp et al., 1995	PM_{10}	0.168	A
	Lower respiratory symptoms (wheeze)	Dusseldorp et al., 1995	PM_{10}	0.061	A
children	Bronchodilator usage	Roemer et al., 1993	PM_{10}	0.078	B
	Cough	Pope and Dockery, 1992	PM_{10}	0.133	A
	Lower respiratory symptoms (wheeze)	Roemer et al., 1993	PM_{10}	0.103	A
all	Asthma attacks (AA)	Whittemore and Korn, 1980	O_3	0.0429	B?
ELDERLY 65+ (14% of population)					
	Congestive heart failure	Schwartz and Morris, 1995	PM_{10}	1.85E-5	B
CHILDREN (20% of population)					
	Chronic bronchitis	Dockery et al., 1989	PM_{10}	1.61E-3	B
	Chronic cough	Dockery et al., 1989	PM_{10}	2.07E-3	B
ADULTS (80% of population)					
	Restricted activity days [2]	Ostro, 1987	PM_{10}	0.025	B
	Minor restricted activity day[3]	Ostro and Rothschild, 1989	O_3	9.76E-3	B
	Chronic bronchitis	Abbey et al., 1995	PM_{10}	4.9E-5	B
ENTIRE POPULATION					
	Respiratory hospital admissions (RHA)	Dab et al., 1996	PM_{10}	2.07E-6	A

Table 4.1 (continued)

Receptor	Impact category	Reference	Pollutant	f_{er} [1]	Uncertainty rating
		Ponce de Leon et al., 1996	SO_2	2.04E-6	A
			O_3	3.54E-6	A
	Cerebrovascular hospital admissions	Wordley et al., 1997	PM_{10}	5.04E-6	B
	Symptom days	Krupnick et al., 1990	O_3	0.033	A
	Acute mortality (AM)	Spix and Wichmann, 1996, Verhoeff et al., 1996	PM_{10}	0.040%	B
		Anderson et al., 1996, Touloumi et al., 1996	SO_2	0.072%	B
		Sunyer et al., 1996	O_3	0.059%	B
	Chronic mortality (CM)	Pope et. al. (1995)	PM_{10}	TBA	TBA

Notes:

[1] The exposure–response slope, f_{er} , is for Western Europe and has units of cases/(yr-person-$\mu g/m^3$) for morbidity, and percentage change in annual mortality rate/($\mu g/m^3$) for mortality.

[2] Assume that all days in hospital for respiratory admissions (RHA), congestive heart failure (CHF) and cerebrovascular conditions (CVA) are also restricted activity days (RAD). Also assume that the average stay for each is 10, 7 and 45 days respectively. Thus, net RAD = RAD − (RHA × 10) − (CHF × 7) − (CVA × 45).

[3] Assume asthma attacks (AA) are also minor restricted activity days (MRAD), and that 3.5 per cent of the adult population (80% of the total population) are asthmatic. Thus, net MRAD = MRAD − (AA × 0.8 × 0.035).

Source: European Commission (1995); Donnan and Hurley (1997).

4.2.2 Particles

There is very substantial epidemiological evidence of adverse acute health effects of particulate air pollution and strong, but much less widespread, epidemiological evidence of chronic health effects. However, ambient particles are a complex mixture, varying in size and in composition, leading to E-R functions in several metrics. It is likely that finer inhaled particles are more toxic per unit exposure ($\mu g/m^3$) than coarser ones. Nevertheless, we have mostly used E-R functions expressed in terms of PM_{10} (particles less than 10 μg in diameter) because of the more comprehensive range of functions available, and because it was judged that results would not depend critically on which PM index was used. Some sensitivity analyses using $PM_{2.5}$ were carried out to check this judgement.

4.2.3 Ozone

The APHEA results have confirmed a relationship of ambient ozone with acute mortality and hospital admissions in Europe, which appears to be additive to that of particles. We give E-R functions expressed in terms of a six-hour daily average. This involves applying conversion factors to functions derived originally using different indices (for example, as one-hour daily maximum; as eight-hour or 24-hour daily average). Data from some of the European APHEA-study cities suggest that six-hour daily average ozone is approximately 0.8 times the one-hour daily maximum. Consequently, E-R functions expressed as a percentage change per unit of one-hour maximum ozone have been scaled upwards by a factor of 1.25 to give the equivalent percentage change per parts per billion (ppb) of a six-hour daily average.

4.2.4 SO_2

In the first ExternE study, any health effects associated with SO_2 as a gas were not quantified (European Commission DGXII, 1995b, chapter 4); the evidence at that time was ambiguous. The APHEA results have, however, established an association of SO_2 with acute mortality, and probably with hospital admissions also, from which E-R functions can be derived. It is still unclear if these associations with SO_2 are causal: it may be that SO_2 is acting as a surrogate for other pollutants, especially fine particles (for example, sulphates) not well quantified in the particle measurements available for study. However, in the APHEA studies the size of the apparent SO_2 effect does not depend on the background concentrations of ambient particles. We therefore have used E-R functions for SO_2 as additive to those for particles and for ozone.[1]

4.2.5 CO

When decisions were made, there were some (but relatively few) epidemiological studies linking ambient CO with acute mortality or (cardiovascular) hospital admissions; and it was unclear if the associated E-R functions were representative or transferable. (Several well-conducted studies which did consider CO did not find a CO-related effect.) Thus, E-R functions for acute mortality and for ischaemic heart disease have been used for sensitivity analyses only. A third E-R function, linking CO with congestive heart failure, was used in the Netherlands, in sensitivity analyses in the UK, but not in the German or Italian implementations. The original E-R functions, expressed as one-hour maximum or eight-hour daily average CO, were scaled up by a factor of 2 or 2.5 respectively, to give values in terms of 24-hour daily averages.

4.2.6 NO_2

European Commission DGXII (1995b) concluded that the epidemiological evidence of acute NO_2 effects was best understood not as causal, but as NO_2 being a surrogate for traffic-related pollution mixtures; and a direct effect of NO_2 was not quantified. (Indirectly, NO_x did contribute, as a precursor, to nitrates and to ozone.) The APHEA results, while reporting positive associations of NO_2 with daily mortality or respiratory hospital admissions in several European cities, also suggest that these associations are highly dependent on background particle levels. Consequently, no direct effects of NO_2 have been used in this study, except for sensitivity analyses. For sensitivity analyses, E-R functions originally in terms of one-hour maximum have been scaled upwards to express results in terms of 24-hour daily average.

4.2.7 Effects in Western and Eastern Europe

APHEA E-R functions linking particles (black smoke) or SO_2 with acute mortality showed higher estimated effects in Western than in Eastern Europe, but did not establish why. There are no corresponding data for other endpoints, or for other pollutants, with which to compare these results. The European functions used in Markandya and Pavan (1999) are from Western Europe, because these are available, and because the countries of principal interest are located there.

4.2.8 Transferring E-R Functions from North America

Where comparisons are possible, they show differences in estimated effects between Europe and North America, raising questions about the transferability to Europe of North American results.

For example, the estimated effect of particles on acute mortality and hospital admissions is generally lower in Europe than in North America. The reasons are not well established; but they may be because of higher co-exposures to SO_2 in Europe. The situation regarding ozone is less consistent. A scaling-down of US functions for use in Europe was considered but not applied.

4.2.9 Conversion from ppb to Gravimetric Units

All E-R functions are expressed in $\mu g/m^3$ except for CO, which is in mg/m^3. The conversion factors used were:

$$1 \text{ ppb } O_3 = 1.997\ \mu g/m^3 \text{ of } O_3;$$
$$1 \text{ ppb } NO_2 = 1.913\ \mu g/m^3 \text{ of } NO_2;$$
$$1 \text{ ppm } CO = 1.165\ mg/m^3 \text{ of } CO.$$

4.2.10 E-R Functions for $PM_{2.5}$

There are not yet enough studies of $PM_{2.5}$ and health to allow quantification of many endpoints and the studies that do exist come almost entirely from North America, raising questions of transferability. Consequently, we have not used the North American functions for $PM_{2.5}$. Instead, for sensitivity analyses we directly converted the E-R functions from PM_{10} to $PM_{2.5}$ by multiplying by 1.67. This conversion factor is based on Dockery and Pope (1994) where it was derived principally from the US experience.

4.2.11 Thresholds

For many of these pollutants, there clearly is a threshold *at the individual level*, in the sense that most people are not realistically at risk of severe acute health effects at current background levels of air pollution. There is however no good evidence of a threshold *at the population level*; that is, it appears that for a large population, even at low background concentrations, some vulnerable people are exposed some of the time to concentrations which do have an adverse effect. On that basis, in this book all health effects have been quantified on a 'no-threshold' basis.

4.2.12 Additivity of Effects

Within endpoints, effects are additive across pollutants unless otherwise stated (see earlier discussion). *Between endpoints for particles*, it was assumed that all respiratory hospital admissions (RHA), congestive heart failure admissions (CHF) and cerebrovascular admissions (CVA) also involve restricted activity days (RAD). In adjusting RADs to take account of hospital admissions, it is arguable whether or not to convert each admission into equivalent hospital days. On balance, we have decided to do so using approximate average length of stay. This gives:

$$\text{Net RAD} = \text{RAD} - (\text{RHA} * 10) - (\text{CHF} * 7) - (\text{CVA} * 45).$$

The final results are insensitive to which adjustment has been made.

Between endpoints for ozone, all asthma attack days (AA) are also minor restricted activity days (MRADs). The function for MRADs refers to adults (80 per cent) only, while that of AA refers to all asthmatics (3.5 per cent). Hence

$$\text{Net MRAD} = \text{MRAD} - (\text{AA} * 0.8 * 0.035).$$

4.2.13 Representing Uncertainty

As noted in the previous chapter labels A to C are used to represent uncertainty on a scale:

A = high confidence, roughly corresponding to $\sigma_G = 2.5$ to 4;
B = medium confidence, roughly corresponding to $\sigma_G = 4$ to 6;
C = low confidence, roughly corresponding to $\sigma_G = 6$ to 12.

4.2.14 Implementation

The various functions can be used more or less directly, by appropriate linkage with incremental pollution and relevant population at risk.

4.3 THE CLASSICAL POLLUTANTS: COMMENTS ON SOME E-R RELATIONSHIPS

The E-R functions used are listed in Table 4.1, with those used only for sensitivity analyses given later, in Chapter 8. The following brief notes may be helpful. They are organised by endpoint.

4.3.1 Acute Mortality

The E-R function for PM_{10} and acute mortality was calculated as the mean of coefficients from Amsterdam (Verhoeff et al., 1996) and Cologne (Spix and Wichmann, 1996), which were similar to high and low estimates respectively from the APHEA study. Estimates for ozone were derived from Barcelona (Sunyer et al., 1996); for SO_2 as the mean of Athens (Touloumi et al., 1996) and London (Anderson et al., 1996).

For sensitivity analyses the E-R function for NO_2 was obtained from the mean of Barcelona (Sunyer et al., 1996) and London (Anderson et al., 1996); for CO from Touloumi et al. (1994) in Athens.

4.3.2 Respiratory Hospital Admissions

The E-R relationship between PM_{10} and respiratory hospital admissions was derived from data from Paris (Dab et al., 1996) within the APHEA study). Data from London (Ponce de Leon et al., 1996) from APHEA provided E-R for ozone, SO_2 and NO_2, the last only included in sensitivity analyses.

4.3.3 Cardiovascular Hospital Admissions

Direct evidence of PM effects on cardiovascular hospital admissions is consistent with the longer-established mortality effects. The E-R function for PM_{10} and cerebrovascular hospital admissions comes from Wordley et al. (1997) in Birmingham, UK. Schwartz and Morris (1995) reported results for Detroit on the relationship between cardiovascular admissions for congestive heart failure and ischaemic heart disease in those aged 65 or more. Exposure-response functions for PM_{10} and CO in relation to admissions for congestive heart failure were derived from this study. Those functions relating to ischaemic heart disease are for sensitivity analysis only.

4.3.4 Emergency Room Visits (ERVs)

Emergency room visits are studied much more in North America than in Europe where (because of differences in health care systems) this endpoint is rare. For that reason, exposure–response functions for ERVs have been used for sensitivity analyses only. The E-R function for Chronic Obstructive Pulmonary Disease (COPD) and PM_{10} was however derived from a European study: Sunyer et al. (1993) in Barcelona. A function for PM_{10} and ERVs for asthma was derived from Schwartz et al. (1993) and

Bates et al. (1990), while that for PM_{10} and ERVs for childhood croup was also European (Schwartz et al., 1991). Relationships for ERVs and ozone are from Cody et al. (1992), with baseline values from Bates et al. (1990), as before.

4.3.5 Restricted Activity Days (RADs)

As noted in European Commission DGXII (1995b), the evidence linking RADs quantitatively with air pollution is among the weakest of all endpoints, because the underlying studies are cross-sectional rather than longitudinal in nature. However no better studies have been published in the meantime. The E-R function for PM_{10} and RADs was therefore based as before on Ostro (1987). Because particle effects estimated in the US may be higher than in Europe, the Ostro estimates were reduced to one-half of their original value. Minor restricted activity days in relation to ozone were derived from Ostro and Rothschild (1989).

4.3.6 Acute Effects in Asthmatics

The health impacts on asthmatics of increased use of medication (bronchodilator usage), increase in respiratory symptoms, and cough were considered in two European studies (Dusseldorp et al., 1995; Roemer et al., 1993) and one US study, Pope and Dockery (1992). The estimate derived from the US has been adjusted downwards to a half of its original value. The exposure–response function for O_3 and asthma attacks is from Whittemore and Korn (1980).

4.3.7 Respiratory Symptoms in the General Population

We have used the results from a study in California, USA (Krupnick et al., 1990) to quantify this endpoint.

4.3.8 Chronic Morbidity in Adults

A PM effect on chronic bronchitis in adults is to be expected, given the evidence on chronic mortality (see below). Direct evidence is however very limited. In European Commission DGXII (1995) we used results from a cross-sectional study by Schwartz (1993) which was unsatisfactory in its linkage with valuation. More recently in ExternE we have followed a recommendation of Ostro, and used an E-R function from Abbey et al. (1995). This gives new cases per year of chronic bronchitis per $\mu g/m^3$ for PM_{10}, and so links more easily with valuation, though there are issues of discounting which have not yet been thought through fully in the present implementation. Also, Abbey et al. is a study in a community of Seventh Day Adventists, with a distinctive lifestyle, and so results may not be transferable.

We considered scaling down the estimated PM effect for application in Europe but did not do so, partly because there are no comparable data on chronic rather than acute effects, partly to compensate for there being no E-R function linking PM with chronic cardiovascular disease.

4.3.9 Chronic Morbidity in Children

The exposure–response functions for chronic morbidity in children have been derived from Dockery et al. (1989). Note however that these are best understood as additional *episodes* of illness, rather than the development of a chronic condition as such, and have been valued accordingly.

4.4 CHRONIC MORTALITY AND PARTICLES

Is there a Causal Relationship?

Mortality effects of long-term exposure to air pollution have frequently been studied using cross-sectional designs, for example in studies which seek relationships between mortality rates and air pollution concentrations in cities. These studies generally reported positive associations for particles. Their major limitation lies in their statistical design; that is, aggregated measures are used for groups of individuals. This approach does not permit control for individual differences in confounders, such as cigarette smoking, leading to doubt about their conclusions. Nevertheless, Lipfert (1994) concluded that, although suffering from methodological flaws, the studies showed consistent associations between particles and mortality at different times and places, and consistent in a positive direction.

Cross-sectional differences in mortality and air pollution have also been studied in prospective cohort studies (Abbey et al., 1991; Dockery et al., 1993; Pope et al., 1995). In these studies the characteristics of the subjects (including where relevant their smoking habits) were collected on an individual basis. Adjustment for confounders at the individual level goes a long way to overcoming the methodological limitations of cross-sectional studies. The positive associations between PM and mortality found in two (Dockery et al., 1993; Pope et al., 1995) of these three key cohort studies is therefore strong evidence of a real effect. The strongest associations were observed for cardiopulmonary disease, with some evidence of a lung cancer effect also. Associations with deaths due to other causes were insignificant.

4.4.1 Choice of Study for Quantification

Of the two principal cohort studies, Pope et al. (1995) is on a much greater scale, involving about 500 000 individuals in 151 US cities, compared with approximately 8 000 people in six cities studied by Dockery et al. (1993). This difference in scale, with its associated reduced chances of confounding, is one reason why we have chosen to base risk estimates on Pope et al. rather than on Dockery et al., despite the better pollution measurements in the latter. Another reason is that Pope et al. give results that are intermediate between the higher estimates of Dockery et al. and the apparent lack of association in Abbey et al. (1991).

Pope et al. (1995) found that, after taking account of confounders, the adjusted mortality rate was 15–17 per cent higher in the most polluted areas compared to the least polluted. Quantification was in terms of sulphates and fine particles ($PM_{2.5}$). These estimates (and those of Dockery et al.) suffer from the limitation that the estimated effects per $\mu g/m^3$ PM are based on relatively recent concentrations, and not on concentrations historically, which arguably are more relevant to the development of chronic disease and to associated premature death. The use of recent concentrations only will have led to over-estimates of the PM effect, on the understanding that PM concentrations have been reduced over time.

The effect of particles as estimated by Pope et al. can be expressed in terms of $PM_{2.5}$ or of sulphates. These can in turn be re-expressed in terms of PM_{10} using the usual conversion factors, as for example in Dockery and Pope (1994). The E-R functions based on $PM_{2.5}$ and on sulphates have however different implications when converted to PM_{10} equivalent: the coefficient derived from $PM_{2.5}$ being about twice that derived from sulphates, expressed per $\mu g/m^3$ PM_{10}. Because of the probable over-estimation from lack of knowledge of historical conditions, and because particle effects are possibly higher in the US (though there are no comparable chronic mortality data), we have based principal results on PM_{10} effects derived from sulphates and used those derived from $PM_{2.5}$ for sensitivity analyses only.

4.4.2 Estimating Years of Life Lost (YOLL)

The years of life lost (YOLL) attributable to air pollution were estimated by linking the regression estimates from Pope et al. (1995) with the populations at risk in the four countries (Germany, Italy, the Netherlands and the UK), using life-table methods. This work was developed and carried out by Brian Miller (Institute of Occupational Medicine (IOM)), following earlier discussions within ExternE. The aim was to evaluate the effect on mortality (YOLL) of a one-year reduction of 10 $\mu g/m^3$ PM_{10}; that is, with concentrations reverting to their original values after one year. This reversion is admittedly an unrealistic scenario in practice; but it permits comparability with other effects evaluated in this volume. Effects on mortality were estimated, using life-table methods, for the currently alive population in each of the four countries. (Because the pollution change was for one year only, there are no effects on as yet unborn cohorts.) Effects were estimated separately for women and for men, using recent (c.1995) demographic data (numbers alive, annual death rates) in one-year age groups, up to age 94 inclusive, the latest years being extrapolated where necessary. The life-table applications involved following the 1995 population right through until everybody now alive had reached 94 years, under two sorts of scenario:

(i) a baseline scenario assuming current hazard rates remain unchanged in future;
(ii) a changed scenario whereby hazards were modified to take account of a pollution change.

Differences in results between (i) and (ii) were considered to be the pollution effect. This was summarised as life years lost, scaled per 100 000 current live population, per 10 $\mu g/m^3$ PM_{10} reduction. Results from all four countries were similar. Consequently, a single set of values was used in the principal analysis. This was 224 YOLL per 100 000 current population per one-year increase of 10 $\mu g/m^3$ PM_{10}, where the hazards were modified using Pope et al.'s results for sulphates. (YOLLs based on scaling using Pope et al.'s results for $PM_{2.5}$ were used in sensitivity analyses, leading to the higher estimate of 470 YOLLs per 100 000 current population per one-year increase of 10 $\mu g/m^3$ PM_{10}.)[2]

4.5 METALS, ESPECIALLY CANCER EFFECTS

4.5.1 Unit Risk Factors

A unit risk factor (URF) is the estimated probability that a person of 'standard' weight of 70 kg will develop cancer due to exposure (by inhalation) to a concentration of 1 $\mu g/m^3$ of a pollutant over a 70-year lifetime. We assume that a one-year increment in exposure increases the lifetime risk of cancer by 1/70 of the URF, that is, by 0.014 of the URF.

4.5.2 Sources of Information

For many metals, there is a range of URFs, calculated in various ways (for example, from epidemiology, using various low-exposure extrapolation models; from animal studies, with animal-to-human extrapolation).

The results of our research show that the estimated effects are low and so are not sensitive to the particular choice of URF used. Generally, the URFs given below are from the World Health Organisation (1987).

4.5.3 General Remarks

Cancer risks can and do vary, by specific compound. URFs and background levels given here are not specific to compound.

Inhalation may not be the only or primary route. The URFs given below are derived from occupational studies with high exposures, where inhalation has been the dominant route. We have not attempted to track the effect of airborne levels through, say, deposition and the food chain.

Principal relevant URFs

Arsenic URF: 3×10^{-3} per $\mu g/m^3$ lifetime exposure $= 4\times10^{-5}$ per $\mu g/m^3$ for one year. Main evidence is for arsenic trioxide (smelter studies) and lung cancer.

Cadmium WHO (1987) does not give URFs; it does not consider the quantification reliable enough.

The US EPA (1985) proposed: URF: 1.8×10^{-3} per $\mu g/m^3$ lifetime exposure $= 2.5\times10^{-5}$ per $\mu g/m^3$ for one year. This was based on US work where co-exposure to arsenic complicated assessment of the evidence of cadmium effects, to the point where the reliability of estimated cadmium effects is unclear. If real, the main cancer endpoint is lung cancer.

Chromium Cancer risk estimates refer to Chromium VI. It seems (WHO, 1987) that 'There are few data accepted as valid on the valency state or availability of chromium in the ambient air'. The conservative approach will be to treat all ambient chromium as chromium VI.

For chromium VI and lung cancer, WHO (1987) gives: URF: 4×10^{-2} per $\mu g/m^3$ lifetime exposure $= 5.6 \times 10^{-4}$ per $\mu g/m^3$ for one year.

Nickel It is unclear if all compounds are carcinogenic. Nickel subsulphides and nickel oxides are the compounds where carcinogenicity is best established, and URFs refer to these compounds. Assuming that all nickel measurements used here are from these compounds, it will give over-estimates of the inhalation effects.

Lung and laryngeal cancers are the sites at risk. Lung cancers will dominate in terms of numbers.

WHO (1987) gives, for nickel refinery dust, URF: 4×10^{-4} per $\mu g/m^3$ lifetime exposure $= 5.6 \times 10^{-6}$ per $\mu g/m^3$ for one year.

Mercury and Lead Neither are carcinogenic to humans.

4.5.4 Latency, Survival Rates and so on for Lung Cancer, and Linkage with Economic Valuation

The following figures are very approximate. Lung cancer has a relatively long latency, that is, time from exposure which triggers the cancer until diagnosis. Assume a latency of 15 years; that is, all effects are deferred for at least 15 years. Survival beyond five years after diagnosis of lung cancer will vary by country but is generally low. If we assume 100 per cent fatality, we will over-estimate the effects, but not grossly so.

In the general population the peak incidence of lung cancer is in the 60–65 year age group. An individual in Western Europe, having attained the age of 60, can expect to live on average about another 20 years. Women live longer than men, on average. On average the life years lost would have occurred, say, 20 to 40 years from the year of exposure. For simplicity, we adjust as if it were 30 years hence.

4.5.5 Non-cancer Risks from Exposure to Mercury

An ambient concentration of 1 $\mu g/m^3$ has been proposed as a conservative threshold for the *direct* (non-cancer) inhalation effects of ambient mercury (WHO, 1987). Ambient concentrations of mercury are generally very low (for example, <10 ng/m^3) compared with this threshold. Any direct health effects can therefore be ignored.

Indirect effects cannot be excluded, for example, through 'deposition in natural water-bodies, resulting . . . in elevated concentrations of methyl mercury in freshwater fish' (WHO, 1987). These effects are however not quantifiable.

4.5.6 Non-cancer Risks from Exposure to Lead

Several health effects have been proposed or examined. The main evidence concerns lead in air and childhood IQ in children. The pathway is in two parts, which need to be considered together:

(a) lead in blood and childhood IQ; and
(b) lead in air and lead in blood.

IQ as an endpoint IQ is a very non-specific measure, and there is controversy about what it really represents. However, relationships with blood lead are better established with IQ, and less variable across studies, than with more specific endpoints.

Lead in blood, and childhood IQ For this part of the E-R function, consider an estimated reduction based on Schwartz (1994) of 2.57 IQ points per 10 μg/dl blood lead in school-age children. For definiteness, assume this refers to school-children say at age 10 years. (In practice, the function is based on studies of school-children over a range of ages.) Assume no threshold.

Lead in air and lead in blood, in children This is not straightforward. Especially in children, ingestion is the principal route of exposure. A recent UK review (EPAQS, 1998) uses a conversion factor of: 1 μg/m^3 in air ~0.5 μg/dl in blood and so ~1.25 IQ points.

Discounting/time of occurrence The adverse effects of exposure to lead will be greatest when the child's nervous system is developing most rapidly. Assume that the biologically relevant exposure occurs over three years, aged 0–2 inclusive. (There will be pre-natal exposure also; and presumably exposure after age 3 may have some effect. So clearly, this is a simplification.) This implies a time-lag of about nine years, between exposure (at 18 months) and effect (at 10.5 years), which is relevant for discounting.

4.5.7 Other Metals

We have not tried to quantify the non-cancer effects of other metals.

4.6 MATERIALS

The analysis of materials remains similar to the previous study shown in Markandya and Pavan (1999). The analysis uses exposure–response functions from the UNECE Integrated Collaborative Programme (ICP) (Kucera, 1994) which has looked at atmospheric corrosion of materials across Europe using a uniform experimental protocol.[3] The functions (based on the four-year exposure programme) for unsheltered stone and metals are shown below.[4] The function for sandstone is used for a mortar attack.

$$\text{Limestone (ICP four years): Mass Loss} \\ = 8.6 + 1.49\ \text{TOW}\ SO_2 + 0.097H^+ \tag{4.1}$$

$$\text{Sandstone (ICP four years): Mass Loss} \\ = 7.3 + 1.56\ \text{TOW}\cdot SO_2 + 0.12H^+ \tag{4.2}$$

$$\text{Zinc (ICP four years): Mass Loss} \\ = 14.5 + 0.043\ \text{TOW}\ SO_2 \cdot O_3 + 0.08H^+ \tag{4.3}$$

The UNECE programme has looked at paint damage, but the four-year function is difficult to implement. There are functions from previous US reviews (Haynie, 1986) and we have used the function for carbonate paint below to look at damages.

$$\Delta ER/t_c = 0.01\ P\ 8.7\ (10^{-pH} - 10^{-5.2}) + 0.006\ SO_2\ f_1 \tag{4.4}$$

It is stressed that there is a higher level of uncertainty with this function, because of the transferability of functions to a European context (different paints show very different acid resistance) and because repainting often occurs for aesthetic purposes. We also note that mass loss may not map into valuation. Materials may be replaced for aesthetic reasons well before physical mass loss dictates they should. In these cases materials damage from pollution may be negligible. This remains a source of controversy but, since no new developments have been made with respect to materials valuation, the same approach as before has been used.

A number of new exposure–response studies have appeared since the air pollution results were quantified. The updated exposure–response functions for materials, based on the eight-year exposure study from the UN/ECE International Co-operative Programme (Tidblad et al., 1998), are now available and will replace the functions above. They include a new function for paint, which will improve the analysis by replacing the US function. In addition, new functions for ozone are now available. A recent research programme has found that there is no evidence in support of a significant effect of O_3 on paints (Holland et al., 1998) in contrast with previous evidence (Campbell et al., 1974; Spence et al., 1975). However, effects were found on rubber products and functions are available. A simplified exposure–response function developed by Holland et al. (1998) for ozone damages has been used in the sensitivity chapter.

4.7 CROPS

The analysis of SO_2 uses the same approach as in Markandya and Pavan (1999). The quantification uses an adapted function from the one suggested by Baker et al. (1986), as shown in equations (4.5) and (4.6). Further details are given in a report of the ExternE Project (European Commission, 1995).

$$y = 0.74(SO_2) - 0.055(SO_2)^2 \quad \text{(from 0 to 13.6 ppb)} \qquad (4.5)$$
$$y = -0.69(SO_2) + 9.35 \quad \text{(above 13.6 ppb)} \qquad (4.6)$$

These functions state that yield will increase with SO_2 from 0 to 6.8 ppb, and decline thereafter. They are used to quantify changes in crop yield for wheat, barley, potato, sugar beet, rye and oats. The crop functions used in this analysis are contained in Table 4.2.

Significant progress has, however, been made with the assessment of damages to crops by ozone. Yield reduction due to ozone was calculated using exposure–response functions for all relevant crops. The functions used were taken from the latest recommendations of the ExternE project based on a survey (Jones et al., 1997, to be published in European Commission, 1998) covering the US National Crop Loss Assessment Network (NCLAN), results reported in earlier ExternE work (European Commission, 1995), and European-based studies (Ashmore, 1993; Fuhrer, 1996). Jones et al. (1997) classified crops into four broad ozone sensitivity

Table 4.2 Crop functions used

Pollutant	Impact	Reference	Remarks
Nitrogen	Add. fertil.	EC (1995)	Benefit, fertilisation effect
Deposition	Needed (kg)		
Acid	Add. lime	EC (1995)	
Deposition	Needed (kg)		
SO_2	Yield loss (dt)	Baker et al. (1986), modified	Considers fertilisation effect of SO_2
AOT40 crops	Yield loss (dt)	Fuhrer (1996)	

categories. These break down crops into four sensitivity classes: tolerant (for example, maize, barley), slightly sensitive (for example, pasture, rye, oats), sensitive (for example, wheat, potato) and very sensitive. Other species which feature in crop production statistics have been added to this list.

For each crop category the 'critical level' has been used in the determination of exposure–response functions, which are assumed to be linearly related to AOT40.[5] The adopted exposure–response functions are shown in Table 4.3. Although there is resistance to the use of these functions in Europe they have been applied in this study, but with these concerns highlighted.

Meat and milk production are assumed to be 50 per cent as sensitive as pasture grass, on which livestock are primarily dependent for food. There is considerable uncertainty in this estimate. In theory, the sensitivity could be anywhere between 0 and 100 per cent, depending upon the existing efficiency of pasture use. However, only 20 per cent of estimated ozone damages to agriculture fall in this category, so the sensitivity of the overall results to alternative assumptions is not large.

An assessment was made of the effects of nitrogen deposition and acid deposition, using the exposure–response functions applied in the ExternE project (European Commission, 1995).

It should be noted that this approach for quantifying crop damages involves some important assumptions. The functions used are largely derived from experiments on a few cultivars under closely controlled laboratory conditions (single pollutant, particular exposure scenarios) and in which other factors of production (notably water) are controlled for growth. It is therefore questionable to what extent results reflect reality in the field, where response may be different for a number of reasons, including:

- sensitivities of different cultivars,
- variable sensitivity across the life cycle,

Table 4.3 AOT40 exposure–response functions

Crop type	Exposure–response function % loss per ppm hour AOT40	Examples
Tolerant crops	0	Maize, barley
Slightly sensitive crops	1.0	Rye, oats
Sensitive crops	1.75	Wheat, potato
Very sensitive crops	3.57	
Meat and milk products	0.5	

- differences in micro-climate and air movement which reduce ozone concentrations in vegetation compared to measured ambient levels,
- interactions with other pollutants,
- adaptation to ozone impacts, and
- humidity and water availability.

The last of these is potentially very important and could result in a systematic error. Water availability is critical to stomatal conductivity, and so to the mechanism by which atmospheric ozone produces leaf damage. In dry conditions, stomata remain closed to reduce water loss. These may well coincide with the hot, sunny and stable atmospheric conditions in which ozone concentrations are highest. In such cases, ozone damage will tend to be lower than laboratory exposure–response functions predict and the values here will be an over-estimate, though this problem is likely to be less applicable on irrigated land and in north-west Europe.

NOTES

1. A US reviewer of this book has pointed out that this remains controversial. We recognise this but feel that European data is more supportive of an additive effect than US data.
2. The first set of calculations of this kind were made by Krupnick et al. (2002) for Canada. Their calculations imply that two hours/person/g/m^3 are 'lost' for a one-year change in PM_{10}.
3. It should be noted that there is some disagreement in scientific circles as to the effects of air pollution on materials which are sheltered and unsheltered. The German and Dutch estimates are based on the assumptions of ExternE, that the damage caused is 50 per cent to unsheltered and 50 per cent to sheltered materials. However, the UK estimates are based on more recent work which shows that there is little difference in corrosion between sheltered and unsheltered materials, and thus do not consider sheltered materials. More research is needed in this area.
4. Where SO_2 and O_3 are concentrations in $\mu g/m^3$, $H+$ = acidity (meq/m^2/year), TOW (Time of Wetness) = fraction of time relative humidity exceeds 80 per cent and temperature > 0°C, $f_1 = 1 - \exp[-0.121$. Relative humidity/(100 − relative humidity)] and t_c = critical thickness loss of 20 μm.
5. AOT40 means Accumulated Ozone Concentration above a threshold of 40 ppbVh.

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5. Developments in valuation

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5.1 INTRODUCTION

Since the publication of the results of the first wave of research (Markandya and Pavan, 1999), a number of developments have been made on the valuation side. These are described below for the areas of crops, health and recreation.

5.2 CROPS

In Markandya and Pavan (1999) it was acknowledged that the method used for valuing crop damage was flawed, because estimated yield losses were taken and multiplied by the unit value of the crops. This takes no account of changes in relative prices consequent upon changes in air pollution. Such changes can result in increases for some crops and reductions for others. What we are really interested in is the change in consumer and producer surpluses for all crops, when pollution levels are reduced to non-anthropogenic levels.

Some work on such changes has been carried out before (see European Commission, 1995). It has not been done, however, for changes in pollution of the kind assumed in the GARP analysis. Hence an exercise was undertaken by the Dutch team to model the pollution changes in a framework where the whole agricultural sector was included and where relative prices were determined as a result of balancing supply and demand. As expected the results are somewhat different from those obtained with the simple partial equilibrium model. The details of the valuation and the estimates obtained are given in Chapter 10. In that chapter a fuller modelling of the agricultural sector reveals that reducing ozone concentration levels is important. Furthermore, economic benefits are unequally distributed between activities, regions and hence also between individual producers. The model estimates a total economic damage of Euro 319 million in 1994 because of ozone pollution to crops. Damages to crops caused by SO_2 seem to be negligible.

A comparison with the simple multiplication method (yield changes valued at fixed prices) reveals that economic adjustments will result in farmers benefiting less from ozone control and consumers benefiting more. As far as total economic surplus is concerned, the difference in magnitude of total economic surplus as calculated with a full economic model and as calculated with the simple multiplication model is relatively modest. This confirms earlier findings.

5.3 HEALTH BENEFITS

5.3.1 Introduction

It is in the area of health benefit estimation that the biggest changes have been made. The main developments have been in the areas of age dependent mortality, and the distinction between acute and chronic mortality. There have also been some changes in the valuation of accidents and morbidity. Each of these is discussed further below. First, however, we consider the basic value of a statistical life, as used in ExternE (European Commission, 1995) and in Markandya and Pavan (1999).

The conventional approach to valuing mortality is based on the estimation of the willingness to pay (WTP) for a change in the risk of death. This is converted into the 'value of a statistical life' (VOSL) by dividing the willingness to pay (WTP) by the change in risk. So, for example, if the estimated WTP is Euro 100 for a reduction in the risk of death of 1/10 000, the value of a statistical life is estimated at $100 \times 10\,000$, which equals one million Euros. This way of conceptualising the willingness to pay for a change in the risk of death makes many assumptions, primarily the implicit assumption that if a risk of death of 1/10 000 is valued at Euro 100, then a risk of death of 1/1000 is valued at Euro 1000. Within a small range of the risk of death at which the VOSL is established this 'linearity' may not be a bad assumption, but it is clearly indefensible for risk levels very different from the one used in obtaining the original estimate.

Estimates of the WTP for a reduction in risk or the willingness to accept (WTA) an increase in risk have been made by three methods. First, there are studies that look at the increased compensation individuals need, other things being equal, to work in occupations where the risk of death at work is higher. This provides an estimate of the WTA. Second, there are studies based on the contingent valuation method (CVM), where individuals are questioned about their WTP and WTA for measures that reduce the risk of death from certain activities (for example, driving); or their WTA for measures that, conceivably, increase the risk of death (for example, increased road traffic in a given area). Third, researchers have looked at actual voluntary expenditures on items that reduce the risk of death from certain activities, such as purchasing air bags for cars.

The ExternE project (European Commission, 1995) and Markandya and Pavan (1999) reviewed 15 European studies of VOSL and concluded that the range of values for the value of a statistical life was between Euro 0.8 and 8.3 million in 1995 prices, with the mean being Euro 4.2 million. It is worth noting that, on average, the highest values come from the labour market studies, followed by CVM studies, with the lowest values emerging from consumer market studies where actual expenditures are involved. By comparison, if we take VOSL from US studies and convert them into Euros, the range is around Euro 2.4–3.6 million for the US studies, against Euro 3.0–5.4 million for a comparable set of European studies (Pearce et al., 1992). Given the higher per capita income in the US compared with that in the countries where the European studies were carried out (mainly the UK), this is somewhat surprising. It is principally the result of higher values in the European CVM studies, particularly the earlier Jones-Lee study (1976) and the Frankel study (1979). Eliminating these two studies, and converting to 1999 prices, produces a range and an average for the CVM group of Euro 2.6–3.6 million and an average across all retained studies of Euro 3.14 million.

By contrast a higher value is proposed by CSERGE (1999). Their recent review of VOSL included 24 CVM studies, 13 consumer behaviour studies and 17 wage risk studies. They concluded the following:[1]

(a) Estimates of VOSL from the CVM studies range from a low of Euro 0.12 million to a high of Euro 15 million, with a mean of Euro 4.75 million.
(b) Estimates from market based studies range from Euro 0.48 million to Euro 3.4 million (no mean given).
(c) Wage risk studies have the greatest variation, with a low of Euro 2.1 million, to a high of Euro 74.2 million, and a weighted mean of Euro 6.5 million.

CSERGE carried out a meta analysis of these studies. A meta analysis involves taking a set of studies and analysing, using statistical methods, the variations in terms of the characteristics of the studies (sample size, mean income of the affected group, baseline risk of fatality, source of fatality and so on). From this analysis they conclude that the best controlled estimate for the UK is about £5.55 million in 1996 pounds, or Euro 8.5 million.

This is much higher than the VOSL value currently in use in the US (where researchers such as Krupnick recommend a value of around \$4 million). One would expect a lower value for VOSL in Europe, where real incomes are lower. Indeed CSERGE suggest, on the basis of their meta analysis, that the income elasticity of VOSL is around 2, meaning that doubling of real income should increase the VOSL by a factor of four. In 1998 (the latest year for which we have data), US per capita incomes were \$29 340. At the same time, the weighted average per capita income of the four countries in this study (Germany, Italy, the Netherlands and the UK), adjusted for differences in the cost of living between the US and the four, was \$20 660. On this basis the income difference is about 40 per cent and the VOSL for the four, implied by the US value of \$4 million and an income elasticity of 2 would be \$2 million, or Euro 2.3 million in 1998 exchange rates (data are taken from World Bank, 2000).

The income elasticity of 2 that CSERGE reports is rather high, in comparison with values from studies involving a wider range of incomes. For example, studies in India (Murty et al., 1992) and elsewhere (Stern, 1977) estimate elasticities of 1–2.

Given the above comparisons, and given that almost all commentators on VOSL in Europe and the US would regard the CSERGE value as too high, we have retained a value of Euro 3.1 million for the VOSL of this study.

5.3.2 Age Dependent Mortality

The problem of age dependent mortality arises because the value of Euro 3.1 million is based on studies in which the individuals involved came from relatively narrow age bands (around 25–55 years, with a concentration in the 35–45 age band). One would expect the VOSL to vary with age, with a possibly lower value for older people, but this does not appear to be supported by the little evidence that is available. Variations of estimated VOSL by age do exist but they do not demonstrate a clear pattern. The one study that provides some evidence on age dependence is Jones-Lee et al. (1985), whose results are displayed in Table 5.1. Other studies, such as Shephard and Zeckhauser (1982), address this question but in a purely theoretical

Table 5.1 Variation in VOSL by age

Age	20	25	30	35	40	45	50	55	60	65	70	75
VOSL as % of age 40	68	79	88	95	100	103	104	102	99	94	86	77

Source: Jones-Lee et al. (1985).

context. Jones-Lee et al. found that VOSL at age 20 is about 70 per cent of VOSL at age 40. Between 20 and 40 it rises slightly, and between 55 and 75 it falls slightly, so that by the age of 75 it is around 77 per cent of the value at age 40. An ongoing EU-funded research project called NEWEXT is investigating these issues, focusing on elderly people.

These are quite small adjustments to VOSL and are, generally, within the margins of error of the estimates anyway. Furthermore, in those cases where we use VOSL, there are other factors that are more important in determining the value that should be applied than this small age dependent variation (such as latency, manner of death, and health status). For all these reasons adjusting VOSL for reason of age dependent mortality is not recommended.

5.3.3 Value of Life Years Lost

Some ExternE results on mortality relate to years of life lost rather than increased probability of death. It is possible to estimate the value of a year of life lost from the estimates of the value of a statistical life, if one has data on the age of the reference group, and some way of estimating the discount factor applied to present versus future years of life. Some work along these lines is cited in the US Fuel Cycle Study (ORNL/RfF, 1994), and indicates that the implied value of a year of life lost, when used to value average years of life lost from premature death caused by cancer, cardiovascular disease and so on, is only around 30 per cent of the value that would emerge from an application of the value of a statistical life.

Pursuing this line of reasoning, one can take a ‘prime age male’ with 37 years of life expectancy and apply the Euro 3.1 million VOSL calculated above. This gives a rough Value of Life Years Lost (VOLY) for such a person of Euro 84 000 (not allowing for any discounting of future years or probability of death in the intervening years). Years of life lost in the case of premature deaths from chronic diseases such as cancer or cardiovascular disease will vary with age of onset and nature of the condition; but an average estimate of 10–15 life years lost may be reasonable. Taking 12.5 years as the mean of that range would give a total value of around Euro 1 million against a VOSL of 3.1 million Euro. The argument is even more compelling for acute effects, where the exposure–response function is picking the impacts of increases in deaths in the days following higher pollution levels. These are generally thought to be affecting persons with short remaining life expectancy. Hence the use of a VOSL, derived from studies of individuals with normal life expectancies, must be questioned there.

What is needed is a framework in which one can use estimates of the value of life years lost in a way that is consistent with the VOSL values reported above. In the next section such a framework is developed. It is fairly general, has been widely discussed, and can form the basis of adjustments to the valuation of some mortality impacts. Ideally values would be derived from original, well-targeted valuation studies. Until these are carried out the method adopted here provides useful first estimates that we believe will give a far more reliable answer than VOSL.

5.3.4 Mortality and Value of Life Years Lost: Acute and Chronic Effects

Acute effects

Discussion within the project team on chronic and acute effects related to air pollution tentatively concluded that for acute effects the loss of life years for those affected is unknown but, in the opinion of medical experts, is likely to be about 0.75 years. For chronic effects no such estimates are available and a different approach is required (see below). The relationship between VOSL and VOLY is taken to be as follows:

$$VOSL_a = VOLY_r \cdot \sum_{i=a+1}^{T} {}_aP_i (1+r)^{i-a-1} \tag{5.1}$$

where a is the age of the person whose VOSL is being estimated, ${}_aP_i$ is the conditional probability of survival up to year i, having survived to year a, T is the upper age bound and r is the discount rate. The above formula assumes that VOLY is independent of age. This assumption is discussed further below.[2]

Estimates of survival probabilities for the EU population are available from Eurostat. The male probabilities have been used to estimate VOLY for individuals aged from 35 to 45. The survival probabilities for all ages and age cohorts for the EU are given in Table 5.2. The rate of discount has been taken to be in the range 0 to 3 per cent. Assuming that VOSL is 3.1 million Euro, the corresponding average value of VOLY for males aged between 35 and 45 is 84 000 Euro with a 0 per cent discount rate and age 35, to 322 392 Euro with a 10 per cent discount rate and an age of 45. These are given in Table 5.3.

Empirical evidence on VOLY and independence of VOLY from age

It is important to compare the value of VOLY above with some direct estimates of this variable. The above analysis has assumed that VOLY is independent of age. This is not critical but it makes the analysis a lot easier. The empirical evidence on the value of VOLY is limited. The one study that has focused on this is Moore and Viscusi (1988). Their methodology is similar to that proposed here and assumes a VOLY that is independent of age. With a VOSL of $6.5 million they derive implicit values of an additional year of life of between $170 000 and $200 000. That makes the VOLY about 2.6 to 3.1 per cent of VOSL. In our case the ratio of VOLY to VOSL is between

Table 5.2 Age cohorts and survival probabilities for males in the EU

Age of cohort	% in pop.	Survival probability	Age of cohort	% in pop.	Survival probability
1	1.176	0.9985	51	1.191	0.9923
2	1.176	0.9988	52	1.191	0.9915
3	1.176	0.9991	53	1.191	0.9906
4	1.176	0.9994	54	1.191	0.9897
5	1.176	0.9998	55	1.191	0.9888
6	1.176	0.9997	56	1.191	0.9872
7	1.176	0.9997	57	1.191	0.9856
8	1.176	0.9997	58	1.191	0.9840
9	1.176	0.9997	59	1.191	0.9823
10	1.176	0.9997	60	1.191	0.9807
11	1.176	0.9996	61	0.974	0.9789
12	1.176	0.9995	62	0.974	0.9771
13	1.176	0.9995	63	0.974	0.9753
14	1.176	0.9994	64	0.974	0.9735
15	1.176	0.9993	65	0.974	0.9717
16	1.176	0.9993	66	0.974	0.9680
17	1.176	0.9992	67	0.974	0.9644
18	1.176	0.9991	68	0.974	0.9608
19	1.176	0.9991	69	0.974	0.9572
20	1.176	0.9990	70	0.974	0.9536
21	1.480	0.9990	71	0.974	0.9485
22	1.480	0.9989	72	0.974	0.9434
23	1.480	0.9989	73	0.974	0.9383
24	1.480	0.9988	74	0.974	0.9332
25	1.480	0.9988	75	0.974	0.9281
26	1.480	0.9987	76	0.974	0.9189
27	1.480	0.9986	77	0.974	0.9097
28	1.480	0.9986	78	0.974	0.9005
29	1.480	0.9985	79	0.974	0.8913
30	1.480	0.9985	80	0.178	0.8821
31	1.480	0.9984	81	0.178	0.8691
32	1.480	0.9983	82	0.178	0.8562
33	1.480	0.9982	83	0.178	0.8432
34	1.480	0.9982	84	0.178	0.8302
35	1.480	0.9981	85	0.178	0.8172
36	1.480	0.9979	86	0.178	0.8001
37	1.480	0.9977	87	0.178	0.7831
38	1.480	0.9975	88	0.178	0.7660
39	1.480	0.9974	89	0.178	0.7489
40	1.480	0.9972	90	0.178	0.7318
41	1.191	0.9969	91	0.178	0.6926

Table 5.2 (continued)

Age of cohort	% in pop.	Survival probability	Age of cohort	% in pop.	Survival probability
42	1.191	0.9966	92	0.178	0.6534
43	1.191	0.9962	93	0.178	0.6143
44	1.191	0.9959	94	0.178	0.5751
45	1.191	0.9956	95	0.178	0.5359
46	1.191	0.9951	96	0.178	0.4287
47	1.191	0.9946	97	0.178	0.3215
48	1.191	0.9942	98	0.178	0.2144
49	1.191	0.9937	99	0.178	0.1072
50	1.191	0.9932			

Note: The figures in the table are the conditional probabilities of survival to year $i+1$, having survived to year i, not the conditional survival probabilities from year a to year i. For someone of age 35 the conditional probabilities are given as: ${}_{35}P_{36} = (0.9981)$; ${}_{36}P_{37} = (0.9981)(0.9979) = 0.9960$; ${}_{37}P_{38} = (0.9981)(0.9979)(0.9977) = 0.9937$ and so on.

Source: EUROSTAT (1995).

Table 5.3 *VOLY for different discount rates (Euro, 1995)*

Discount Rate (%)	VOLY with age 35	VOLY with age 45
0	84 100	111 600
3	141 100	168 500
10	301 400	322 400

Note: Corresponding value of VOSL is 3.14 million Euro.

Source: Authors' calculations.

2.6 and 10.2 per cent, with the higher ratio values corresponding to the 10 per cent discount rate and the lower values to the 0 per cent discount rate. Moore and Viscusi derive VOLY from econometric estimates in which expected life years lost appear as a specific variable. The latter are constructed on the basis of the risk of death and the discounted life years remaining. The two methods are not directly comparable but have the same flavour and, encouragingly, come up with similar answers in terms of the relationship between VOSL and VOLY.

A more recent study in the Value of Life Years Lost (VOLY) vein is that of Johannesson and Johansson (1996). They use the contingent valuation

Table 5.4 WTP for one year of life at age 75 and corresponding values for one year of life immediately

Age of payment	WTP for one life year at 75 (Euro)	WTP for one life year now (Euro)	
		At 11% discount	At 3% discount
18–34	1400	149 352	5960
25–51	1760	59 830	5250
52–69	2020	8040	3100

Note: 1 Euro = 8.7 SEK in 1995 (year of study). Estimates of current WTP are for the middle of the age range.

Source: Derived from Johannesson and Johansson (1996).

method to look at the WTP of different respondents aged 18–69 for a device that will increase life expectancy by one year at age 75. The precise question posed to people is:

> The chance of a man/woman of your age reaching 75 years is x percent. On average a 75 years old person lives for another ten years. Assume that if you survive to the age of 75 years you are given the possibility to undergo medical treatment. The treatment is expected to increase your expected remaining length of life to 11 years. Would you buy this treatment if it costs C kroner and has to be paid for this year?

In total 2455 individuals were interviewed, of whom 82 per cent responded to the questionnaire. The replies were analysed using the logit model for binary choices. The results obtained are reported in Table 5.4, converted into Euro.

These figures are generally much lower than those of Moore and Viscusi. An implicit discount rate can be computed from the values and this comes out at between 0.4 and 1.3 per cent, which is much lower than the rates estimates in other studies (Cropper et al., 1994, Viscusi and Moore, 1989).

The above discussion reveals that, although there is a move towards using VOLYs in the literature, more work needs to be done before a first set of estimates are arrived at. For the present, it is recommended that equation (5.1) be used to estimate the costs of air pollution via mortality for all cases, using the values of VOLY as given in Table 5.3 and replacing the normal conditional survival probabilities with the probabilities associated with the particular case. For acute effects this calculation is relatively simple. Assuming that individuals are affected immediately, and life expectancy is reduced to 0.75 years, the P_is are replaced with 1 for the following 0.75 years and zero thereafter. This results in an estimated mortality cost of: Euro

73 500 (0 per cent discount rate), Euro 116 250 (3 per cent discount rate) and Euro 234 000 (10 per cent discount rate). Note, however, that the exposure–response function for chronic effects of air pollution implicitly covers reduced life expectancy from acute effects as well as chronic. Investigation of the sensitivities of different approaches suggests that quantification based solely on the recommendations made below for chronic effects will give an acceptably accurate result.

Chronic effects

For chronic effects the calculation is more complicated. The model followed here is as follows. Once exposed, impacts can occur with a latency that is variable, but could be as much as 50 years. Once the impact is under way, survival probabilities are altered to an extent that is undetermined. Although there are great uncertainties, the study team have estimated, for representative populations, the number of years of life lost as a result of an increment in the hazard in year *i*, in each future year. Call this $YOLL_i$. We have also estimated for such a case the total number of years of life lost in the population ($YOLL_{tot}$). The value to be attached to a case of chronic mortality can then be estimated as $VOLY_{chronic}$. The formula proposed for this is:

$$VOLY^r_{chronic} = \sum_{i=1}^{i=t} \frac{YOLL_i}{YOLL_{tot}} \times \frac{VOLY^r}{(1+r)^{i-1}} \tag{5.2}$$

There will be different values for chronic illnesses, depending on the values obtained for $YOLL_i$ and $YOLL_{tot}$, and of course on the value of r. Equation (5.2) has been used to estimate the values for different latency effects. The results are reported in Table 5.5. The age distribution of the population is that of the European Union and the survival probabilities are those for Germany. Note that the last case, where latency and risk are spread out evenly over 30 years, is the one considered appropriate for chronic mortality arising from airborne particulate matter. This is discussed below.

Finally, there has been considerable discussion on the valuation of chronic impacts in the exposure–response functions by Pope et al. (1995). The issue is that the equations provide an estimate of the years of life lost, not the number of excess deaths. As far as the valuation is concerned, we should value years as described in equation (5.2). The relevant valuations for chronic mortality are those in Table 5.5, in which latency and risk are uniformly distributed over 30 years. In Table 5.5 the figures are given separately for males and females. Taking a weighted average of 51 per cent female and 49 per cent male, the numbers derived for chronic mortality in the case of PM exposure are: Euro 98 000 (0 per cent discount rate), Euro 84 330 (3 per cent discount rate) and Euro 60 340 (10 per cent discount rate).

Table 5.5 $VOLY^r_{chronic}$ for different latencies and discount rates (Euro)

	Discount rate		
	0%	3%	10%
$VOLY^r$	98 000	155 000	312 000
Latency 0 years			
female	98 000	122 123	169 198
male	98 000	120 187	162 059
Latency 30 years			
female	98 000	53 820	11 175
male	98 000	53 810	11 133
Latency and risk distributed over 30 years			
female	98 000	84 680	61 269
male	98 000	83 969	59 371

Table 5.6 Estimates of mean years of life lost (YOLL) and subsequent valuation of fatalities, in Euro, for cases of different types of cancer

Type of cancer	Leukaemia	Lung cancer	Stomach cancer	Nasal cancer
Causative pollutants	benzene, butadiene	PAHs, diesel particulates	ethylene oxide	formaldehyde
Latency (l) in years	8	15	15	20
Estimated mean YOLL	22	16	15	14
Discount rate				
0% (VOLY = 98 000)	2 160 000	1 570 000	1 470 000	1 370 000
3% (VOLY = 155 000)	1 810 000	1 080 000	992 000	848 000
10% (VOLY = 312 000)	1 180 000	418 000	373 000	252 000

Note: The cost of illness (estimated at Euro 450 000) should be added to these values (see Table 5.5).

For cases with a long latency period it is appropriate to add a cost for the period of pain and suffering in addition to the values of life years lost that have been reported above. This is dealt with under morbidity values, in the next section.

A similar analysis has been done for a number of cancers. The figures are given in Table 5.6. In this work estimates were made of latency, years of life lost and survival time after diagnosis. The method by which survival probabilities are estimated is a matter of some discussion. To go into any further detail is, however, outside the scope of the valuation framework. It should also be noted that these figures are being revised, as more accurate data on these epidemiological parameters are being included.

To summarise, the mortality values used in this study are:

For VOLY in general:	Euro 98 000 at discount rate 0 per cent; Euro 155 000 at discount rate 3 per cent; Euro 312 000 at discount rate 10 per cent.
For chronic effects in general:	as given in Table 5.5.
For different types of cancer:	as given in Table 5.6.

5.3.5 Morbidity Impacts

There is an enormous US literature on valuing morbidity effects, and a virtual absence of one in Europe, although the situation has improved in recent years (see, for example, CSERGE, 1999). Given the collaborative nature of this study, maximum use has been made of the excellent work carried out in this area by the US team, with modifications to their findings as and when appropriate.

The WTP for an illness is composed of the following parts: the value of the time lost because of the illness, the value of the lost utility because of the pain and suffering and the costs of any expenditures on averting and/or mitigating the effects of the illness. The last category includes both expenditures on prophylactics, as well as on the treatment of the illness once it has occurred. To value these components researchers have estimated the costs of illness, and used CV methods as well as models of avertive behaviour.

The costs of illness (COI) are the easiest to measure, based either on the actual expenditures associated with different illnesses, or on the expected frequency of the use of different services for different illnesses. The costs of lost time are typically valued at the post-tax wage rate (for the work time lost), and at the opportunity cost of leisure (for the leisure time lost). Typically the latter is between one half and one third of the post-tax wage. Complications arise when the worker can work but is not performing at his full capacity. In that case an estimate of the productivity loss has to be made. It is important to note that COI is only a component of the total cost. However, since the other components are difficult to measure, estimates have been made of the relationship between the total WTP and COI. Rowe et al. (1995) review these studies which give ratios of 1.3 to 2.4 for WTP/COI and recommend a ratio of 2 for adverse health effects other than cancer and 1.5 for non-fatal cancer.

CVM is the only approach that can estimate the value of the pain and suffering. The difficulties are those generally associated with the use of CVM and, in addition, of allowing for the fact that it is difficult to know which of the many costs are included in the given responses. In general, respondents will not include those costs that are not borne by them as a result of the illness (for example, medical insurance). In that event, such costs need to be added. In this category one should also include the cost, in terms of pain and suffering, that the illness causes to other people (the so-called altruistic cost).

Avertive behaviour is the most complex of the three to model. It involves the estimation of a health production function, from which one would be able to estimate the inputs used by the individual in different health states, and, taking the difference in value between these, obtain the cost of moving from one health state to another. The difficulty is in estimating that function, where many 'inputs' provide more than one service (for example, bottled water, air conditioners), and where the changes in consumption as a function of the state of illness are difficult to estimate.

The broad groups under which the estimates can be classified are as follows:

(a) estimation of restrictive activity days;
(b) cost of chronic illnesses;
(c) valuation of symptom days;
(d) estimation of altruistic costs.

Restricted activity days (RADs)

A large number of studies, using COI as well as CVM methods, have been used to estimate several categories of RADs. These are differentiated by illness (respiratory RAD (or RRAD), angina RAD and so on), and by severity of impact (minor RAD (MRAD) versus 'normal' RAD). It is stated that these impacts are among the easier of the health impacts to value, as they relate to acute events, lasting a well-defined period. The US study provides central or best estimates for these impacts which can, as a first approximation, be taken in the European study using a purchasing power parity (PPP) exchange rate. Although there may be grounds for arguing that medical costs are somewhat higher in the US (even at the PPP rate), the errors involved in transferring the estimates are likely to be dwarfed by those arising from other sources. Until a corpus of European studies is available, therefore, it is recommended that these values for RAD be taken from the US study and an inflation adjustment be applied to the US value. This gives a value of Euro 75 per RAD.

The US study (ORNL/RfF, 1994) points out that the central values provided for an RAD cannot be simply multiplied by the number of days lost, because one would expect that the value of each additional day declines as one loses more days, and indeed the empirical evidence supports this (the average value declines as the number of days lost increases). The conclusion that the estimated rate of decline in the value of an RAD with the number of days is too inaccurate to be of use is probably correct, making the use of a single value the best course to follow at this stage.

Chronic illness

Chronic illness endpoints (broadly interpreted as impacts that occur over a long period of time) include the following:

- chronic bronchitis in adults;
- non-fatal cancers and malignant neoplasms;
- chronic cases of asthma.

For *chronic bronchitis* in adults CVM studies in the US suggest a value of Euro 240 000 (Krupnick, 1998). This value was updated in the course of the project to take account of the latest developments (it used to be Euro 105 000).

Non-fatal cancers are valued on the basis of direct costs of illness plus foregone earnings (COI). Average values come out at Euro 0.30 million. There is no WTP estimate for this, although, as noted above, there are studies suggesting that WTP is equal to 1.5 times COI for non-fatal cancers. On this basis the value for non-fatal cancers should be Euro 0.45 million. Note that this value should be added to the mortality costs of cancer, where the case is better described as a fatal cancer with a long latency period and a substantial survival period.

Chronic cases of asthma have not been valued in direct empirical studies and we therefore approximated these by using the value of chronic bronchitis according to old studies, that is, Euro 105 000.

Symptom days

The US Fuel Cycle Study reviews the extensive literature on the valuation of symptom days. These include CVM studies, as well as some that combine avertive behaviour and CVM. Although the work carried out is impressive, there are unfortunately still many difficulties to be resolved. The CVM studies have problems of low response rates and extreme bids that have to be discounted. There is also a difficulty in knowing the extent to which the responses include the use of avertive measures. Some of the results appear to indicate that the latter are not always allowed for. Some studies (Dickie et al., 1986, 1987) have included information on actual avertive behaviour and revealed responses but, as the US Fuel Cycle Study points out, ‘the results of this study need considerable refinement before they can be used with confidence in a morbidity benefit analysis. The limitations arise in the theory, data, statistical, and implementation phases of the study'.

The most recent study of this type in Europe was provided by Navrud (1997). The author performed a CV study of a representative sample of 1000 Norwegians (above 15 years) to estimate their WTP to avoid one and 14 additional days annually (sub-samples A and B respectively) of seven 'light' health symptoms and asthma. The seven light symptoms were: coughing, sinus congestion, throat irritation, eye irritation, headache, shortness of breath and acute bronchitis. First, people were asked about how many days they had experienced having each of these symptoms in the last 12 months; then they were asked what costs this implied in terms of medicines, hospital visits and lost time at school, work or leisure. Finally, they were asked about their maximum WTP to avoid having one or 14 more days of each symptom the next year, compared to what they had experienced last year.

Air pollution was not mentioned in the survey; thus the values obtained are non-contextual. The numbers are believed to be more transferable from one type of project to another, and possibly between countries, but they do not relate specifically to air pollution problems.

In general, however, the state of the art is not yet at a stage where symptom days can be valued with any confidence (even by the less rigorous standards that apply to environmental valuation as a whole). Nevertheless, given the availability of one plausible value, it is better than excluding the endpoint altogether, and a value of Euro 7.5 (1995 prices) per symptom day (see ORNL/RFF, 1994) has been included in the analysis. This has been applied to both symptom days and chest discomfort days. In the valuation of a symptom day, covering the range of symptoms described above, Navrud gets values of Euro 2.5 to 7.5. The value of a symptom day in ExternE (1995 prices) is Euro 7.5, which is at the upper end of the range from Navrud.

Altruistic impacts

As with symptom days, estimates of the impact of an illness on the utility of others is not sufficiently developed to be used in a valuation exercise. One US study (Viscusi et al., 1988) came up with an altruistic value for each case of poisoning avoided of more than five times the private valuation. The experiment consisted of a CVM in which individuals were asked their WTP for a TV campaign that would reduce poisoning resulting from poor handling of insecticides. However, the study had a relatively unsophisticated design and the results need to be confirmed in other studies. Work in the UK by Needleman (1976) and Jones-Lee et al. (1985) has suggested that the altruistic values are around 40–50 per cent of the private total valuations.

Again, however, these are isolated findings and need to be corroborated. In view of the current state of the art in this area, altruistic valuations have not so far been included in the ExternE study. However, the omission of altruism provides a bias to underestimation of damages.

There are a number of further endpoints that have been valued. The most recent European evidence is from Navrud (1997) and CSERGE (1999) with Rowe et al. (1995) providing a review of the more recent US studies.

Emergency room visits (*ERV*) The WTP to avoid an ERV is estimated at Euro 223. This has been taken to value ERVs generated as a result of concentrations of PM_{10} and ozone.

Respiratory hospital admissions (*RHA*) The WTP to avoid an RHA is estimated at Euro 7870. This value has been taken to value RHAs generated as a result of concentrations of PM_{10} and ozone. It should be noted however that there remains considerable uncertainty in this valuation. For example, another study (CSERGE, 1999) derives a lower value of Euro 1840 – the WTP element being Euro 468.

Children's bronchitis and cough An estimate is made for acute bronchitis in children based on COI data and using a COI/WTP ratio of 3 to obtain a value of Euro 225 per case.

Asthma attacks The WTP to avoid each attack is estimated at Euro 37 – adjusting up from COI estimates. It applies to the impacts of concentrations of PM_{10} and ozone. This compares to values of Euro 15 and Euro 45 in the Navrud study, for adults and children respectively.

Severe hereditary effects Again there are no WTP estimates of such impacts. Given the seriousness with which such effects are viewed, they have been valued at the same as a statistical life, Euro 3.1 million.

Table 5.7 gives a summary of the valuation of the morbidity endpoints discussed above, including values for accidents that are discussed further below.

5.3.6 Conclusions

This section has reviewed the literature on the valuation of the health impacts of the fuel cycles. Health impacts are probably the most important of all to value and also the most difficult conceptually. The section began by looking at the methodological issues arising in the valuation. Impacts to

Table 5.7 Valuation of morbidity endpoints in this study (1995 prices)

Endpoint	Value (Euro)	Estimation method and comments
Acute morbidity		
Restricted activity day (RAD)	75	CVM in US estimating WTP
Symptom day (SD) and minor restricted activity day	7.5	CVM in US estimating WTP. Account has been taken of Navrud's study
Chest discomfort day or acute effect in asthmatics (wheeze)	7.5	CVM in US estimating WTP. Same value applies to children and adults
Emergency room visits (ERV)	223	CVM in US estimating WTP
Respiratory hospital admissions (RHA)	7870	CVM in US estimating WTP
Cardiovascular hospital admissions	7870	As above
Acute asthma attack	37	COI (adjusted to allow for difference between COI and WTP). Applies to both children and adults
Chronic bronchitis in adults	240 000	CVM study in US
Non-fatal cancer	450 000	US study
Malignant neoplasms	450 000	These are valued as being similar to non-fatal cancer
Chronic case of asthma	105 000	Based on treating chronic asthma as new cases of chronic bronchitis using old values for bronchitis
Cases of change in prevalence of bronchitis and cough in children	225	Based on COI, adjusted for COI/WTP ratio
Severe hereditary effect	3 140 000	Perception of severity of effect as equivalent to mortality. Hence VOSL used
Occupational injuries (minor)	77.5	French compensation payments
Occupational injuries (major)	22 600	French compensation payments
Workers & public accidents (minor)	6970	TRL (1995). New estimates
Workers & public accidents (major)	95 050	TRL (1995). New estimates

be valued were divided into mortality, morbidity and accidents. For mortality impacts there are several issues that need to be resolved but, at the present time, a value of Euro 3.1 million in 1999 prices appears to be the best central estimate. A range of problems that need to be addressed in the future was identified. VOSL should not be used in cases where the hazard has a significant latency period before impact, or where the probability of survival after impact is altered. In such cases the value of life years lost (VOLY) is recommended. The values for VOLY are Euro 100 000–300 000 depending on the discount rate and life expectancy. For specific latency periods and chronic illnesses that alter the survival probabilities estimates are given in Tables 5.5 and 5.6.

For morbidity, categories of impacts were divided into estimates of restrictive activity days (RADs) and variants of that; estimates of chronic illness, symptom days, and altruistic impacts. For RADs and variants, US estimates have been used as one source, with conversion into Euro made using a purchasing power parity exchange rate. In addition, recent European studies have been reviewed, and estimates taken from them as appropriate. In some cases no WTP estimate is available, either from the US or from Europe. In such cases a cost of illness (COI) approach has been used, with the COI value being grossed up for items not captured by that method (that is, pain and suffering). An estimate of the grossing-up factor was provided. Estimates of chronic illness can be made via two routes.

5.4 RECREATIONAL AND AMENITY BENEFITS

Water damage costs were addressed for the first time in this research project. The analysis provided first estimates of the types of recreational losses that are likely to result from a fall in water quality as a result of anthropogenic activity. Angling was the particular form of recreational activity investigated and this was undertaken in the context of a case study for the UK alone. The monetary values adopted were derived from a study commissioned by the UK Foundation for Water Research[3] which classified water quality into three broad categories: good, medium and poor and for different fish types. The descriptions of these categories are set out in Table 5.8.

These values are considered by the project team to have good validity and general robustness. In order to reflect different levels of water quality more accurately, intermediate values were interpolated between the three categories of values given in the original contingent-valuation study. This interpolation was done using a linear function. There is no evidence that this is not a broadly reasonable assumption to make though it is something that requires further investigation as water accounting methodologies develop.

Table 5.8 Categories of water quality and fish types

Categories: Salmon

Category	Description
Good	1 in 10 chance of catching a salmon for each day's fishing by an average angler
Poor	1 in 40 chance of catching a salmon for each day's fishing by an average angler

Categories: Game (e.g. Trout)

Category	Assumed average density of > 20 cm fish
Good	> 2 fish/100 m^2
Medium	0.8 to 2 fish/100 m^2
Poor	< 0.8 fish/100 m^2

Categories: Coarse

Category	Assumed average biomass of coarse fish (except eels)
Good	> 2000 g/100 m^2
Medium	600–2000 g/100 m^2
Poor	< 600 g/100 m^2

Transferability was not seen to be a problem since the results were obtained from a country-wide survey. Clearly, this would become an issue were these values to be used in a wider European context. This needs to be addressed in future work on this topic.

Water amenity value had to be investigated in a rather crude way because of data and resource constraints. The exercise looked at the relationship between residential property prices and their waterfront locations. Thus, the value reached simply identified how much the existence of a water-course in proximity to their property was worth to the property owners. This value can only be related to variations in water quality if one assumes that a certain level of (presumably) poor water quality in a water-course can be equated to the non-existence of that water-course. Any price premium that is found to exist on the waterfront properties may then be interpreted as an implicit valuation of the existing level of water quality. The exercise in Chapter 12 adopts a price premium range derived from UK-based surveys and this is used to calculate an aggregate value interpreted as a valuation of existing water quality above some poor level that is equated to the water's non-existence.

This exercise does not value the water amenity damage costs; rather, it simply signals that amenity is a potentially important source of value when considering water resources and damages. Future work on this subject would seek to use more disaggregated-level data which could identify different price premiums for different water quality levels. This may enable us to use the premium from a stretch of very good water quality as a proxy for 'pristine' condition and to calculate the damage by subtracting values from less good stretches of water from this value.

NOTES

1. An exchange rate of \$1.28 to the Euro (ECU before 1999) has been taken (IMF, International Financial Statistics).
2. Note that the formula is set up to allow for the possibility that VOSL is age dependent. Given information on the ages of individuals affected by air pollution the estimates of costs can be adjusted for this factor as well, but this has not been done in the present analysis. The analysis has been criticised for this assumption. However, the team feel that, as a first approximation, it is not bad and is better than the alternative of using a VOSL figure.
3. Foundation for Water Research – FWR (1996).

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6. The methodology for the estimation of impacts and damage costs caused by ambient air pollution

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6.1 INTRODUCTION

In this chapter the methodology used for the estimation of impacts and damage costs caused by air pollution and their attribution to sources is described. The impact pathway approach which was developed in ExternE, improved in GARP I (European Commission, 1995) (see Chapter 3) and already used for many studies, for example, European Commission (1997), was followed for the calculations. Measured as well as modelled concentration data were used to estimate physical impacts and damage costs on human health, building materials and field crops by applying the recommended exposure–response functions and monetary values discussed in Chapters 4 and 5 respectively. While the use of measurement data allows the estimation of impacts resulting from the actual concentration load within a region, models can be used to trace back pollutant transport and thus to establish a link between impacts and a specific source at a specific site. The two following sections briefly describe the two different methodologies.

6.2 ESTIMATION OF IMPACTS AND DAMAGE COSTS BY USING MEASURED CONCENTRATION DATA

In order to obtain the concentration levels of pollutants at the receptors' locations, annual average values of measured concentration data for Germany, Italy, the Netherlands and the UK are applied to derive pollution maps. These are used together with the spatially disaggregated receptor data which are described in Chapter 8, Subsection 8.2.3, to estimate physical impacts and damage costs for a one-year exposure to pollutant concentrations, including human health effects, damages on building materials and yield losses for field crops.

The current study covers damages caused by the pollutants SO_2, NO_2, PM_{10}, ozone, CO, some organic compounds and as far as possible impacts caused by heavy metals in aerosols. The calculations were carried out for the year 1994 (Germany, Italy and the Netherlands) and 1996 (UK). Additionally, year on year changes between 1990 and 1995 were analysed for the UK. The pollutants considered and the spatial resolution of the pollution maps vary between countries, because the available databases are different. In Sections 8.2 and 8.4 the respective data sets are described in more detail.

6.3 ATTRIBUTION OF DAMAGES TO ANTHROPOGENIC SOURCES

Measurement data show the actual concentration load within a geographical region, but for some pollutants natural emission sources significantly contribute to the total concentration level. In the impact analysis only the pollution load resulting from anthropogenic emissions is relevant. Therefore, the impacts are estimated on the basis of the differences between the measured total concentration levels and natural background levels. A detailed discussion of the natural background levels used for the calculations can be found in Section 8.2. Because of the large uncertainties related to the definition of natural background concentrations, a sensitivity analysis was performed to take account of different assumptions.

6.4 ATTRIBUTION OF DAMAGES TO SOURCE COUNTRIES AND SECTORS

For the analysis of the linkage between economic activities and environmental damages through air pollution, the EcoSense model, an integrated tool for environmental impact analysis and the assessment of external costs, was used. Various databases are linked to the program system, which is used together with air quality models for the estimation of physical impacts and damage costs caused by different emission sources. The databases include emission data, meteorological data, population distribution, crop production data, material inventories, exposure–response functions and monetary values.

In the following the emission database, the chemical dispersion models and the calculation procedure applied in the current analysis are briefly described. Further information concerning the remaining databases can be found in Chapter 8.

6.4.1 Emission Database

For the attribution of damages to economic sectors and countries of origin the CORINAIR 1990 emission database is linked to the system and used in the analysis. It includes emission data of the substances NO_X, SO_2, NH_3 and VOC all over Europe, aggregated to economic sectors (SNAP = Selected Nomenclature for Air Pollution) and administrative regions (NUTS = Nomenclature of Territorial Units for Statistics). Additionally, an emission map for particles (TSP) in Germany was derived from total emissions of economic sectors in Germany by using the German population distribution. So the damage calculation covers impacts from the primary pollutants mentioned above and the related secondary pollutants, which are sulphate and nitrate aerosols and ozone.

6.4.2 Air Quality Modelling

The air quality models applied to the damage estimations are the Windrose Trajectory Model (WTM) and the Source Receptor Ozone Model (SROM). The WTM is a user-configurable trajectory model based on the Windrose approach of the Harwell Trajectory Model developed at Harwell Laboratory, UK (Derwent et al., 1988; Derwent and Nodop, 1986). Within this study the model is used to assess concentration increases of primary emitted particles as well as of secondary particles (nitrates and sulphates) and SO_2. Additionally, dry and wet depositions of nitrogen, sulphur and acids were modelled with WTM. The chemical mechanism followed for the analysis is the same as applied in the ExternE studies. Details can be found in European Commission (1999) and Trukenmüller and Friedrich (1995).

The SROM is a model which estimates ozone concentration by using source–receptor matrices. These were derived from the results of the Cooperative Programme for Monitoring and Evaluation of the Long-range Transmission of Air Pollutants in Europe (EMEP) Ozone Model for different reduction scenarios (Simpson et al., 1997). The model is based on the EMEP iteration model developed by David Simpson (Simpson and Eliassen, 1997).

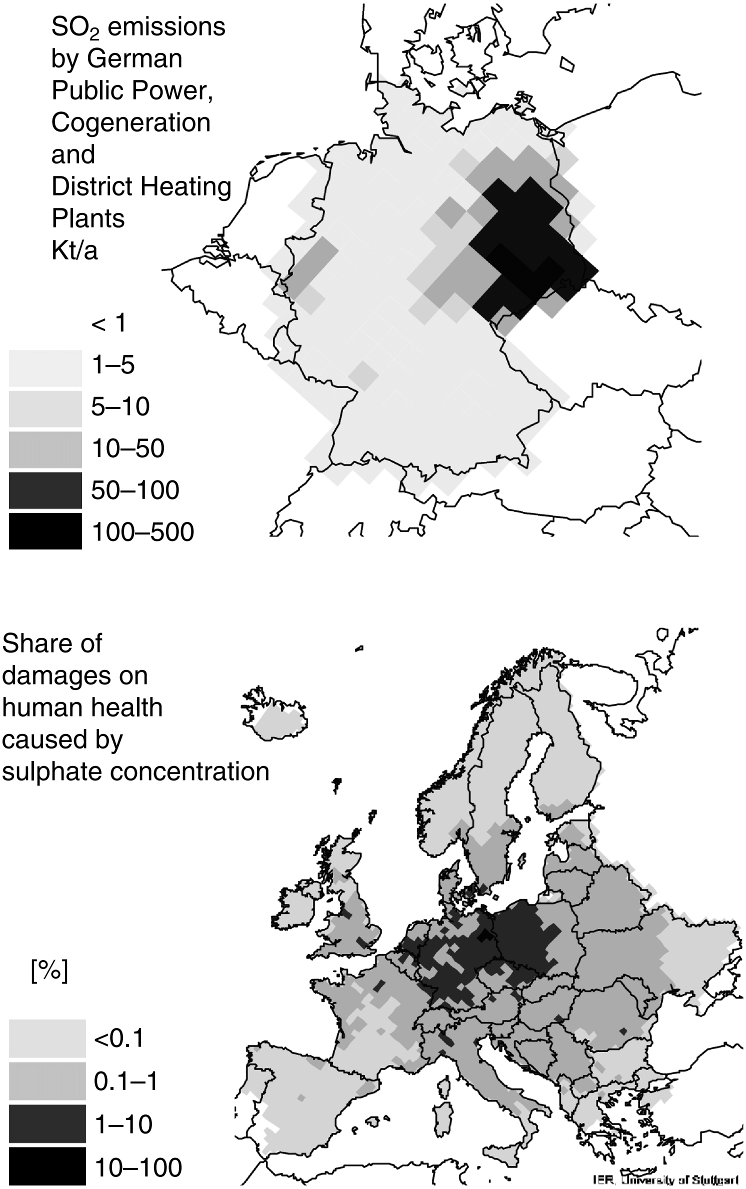
6.4.3 The Procedure of Damage Estimation

The first step in estimating the damages caused by emissions of an economic sector within a given geographic region of Europe is to create an emission scenario. To do this, the CORINAIR 1990 emissions are reduced by the respective sector's emissions in the relevant region. The emissions assigned according to Nomenclature of Territorial Units for Statistics (NUTS) are then distributed to the regular EMEP50 (50×50 km^2) grid and air quality modelling is carried out. The resulting concentrations are subtracted from the respective concentrations estimated for the CORINAIR 1990 emission scenario. The differences in the concentration levels show the concentration load caused by the emissions of the respective economic sector in 1990. The concentrations are used together with receptor data, for example, population distribution, to estimate physical impacts and damage costs. Thus, beside the total damages caused by the emissions of the respective economic sector, the damages occurring within the various countries are also assessed. As an example of the application of the methodology, the estimation of health damages caused by electricity generation in Germany is shown in Figure 6.1.

6.4.4 Damage Estimations Covered by the Current Study

The methodology described is used to estimate physical impacts and damage costs caused by emissions due to all economic activities in each country of the EU-15 states and economic activities in the main economic sectors of the CORINAIR nomenclature (SNAP1) within the four countries (Germany, Italy, the Netherlands and the UK). For this purpose, different kinds of emission scenarios are built. Alongside scenarios in which the emissions of one CORINAIR main sector in a country are set to zero, emission scenarios in which the emissions of all economic sectors in one country except the sector 'Nature' are reduced to zero are applied for the damage calculations.

The possibility of locating the regions in which the damages occur is used to estimate the share of damage costs caused inside and outside the respective countries. Thus, the export of damages is assessed for all EU-15 countries and for each of the individual CORINAIR main economic sectors of Germany, Italy, the Netherlands and the UK. Additionally, the damages caused by all member states within the individual EU-15 countries were estimated. By aggregating the damage costs caused within each country, damage imports from the remaining EU-15 member states were derived. These were used to identify 'net importers' and 'net exporters' of pollution related damages within the European Union. The results are presented in Chapter 9 together with an analysis of linearity and a comparison of modelled pollutant concentration with measured pollution data.



Source: These are original figures based on CORINAIR data distributed to the EMEP 50 Grid.

Figure 6.1 *SO_2 emissions and shares of damages caused by the German sector 'Public Power, Commercial and District Heating'*

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7. Defensive expenditures

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7.1 INTRODUCTION

This chapter outlines the methodological issues that arise when defensive expenditures are considered in an accounting framework. The principal approaches that have been developed to devise such a framework are described and then the approach taken here is explained.

It has long been recognised that the broad concept of defensive expenditures could be an important indicator to use in conjunction with the figures presented in conventional economic accounts. Since the mid-1970s, attempts have been made to develop series data on pollution abatement and other environmental expenditures. The issue of defensive expenditures has gained increased momentum in both academic and government policy circles in recent years and quantification of defensive expenditures is considered to be crucial for addressing environmental policy. This is particularly the case since these estimates can be used both in the context of environmental monetary accounts and of response indicators.

An insight into the overall purpose of recording such expenditures is provided by Peskin (1999), who draws a distinction between the two basic functions of accounting, *scorekeeping* and *management*. Scorekeeping essentially maintains a record of performance that is represented in the form of aggregate statistics. Some commentators would like to see estimates of defensive expenditures used in this way, with the aim of deducting the figures to arrive at measures of 'green GDP'. The position taken by the project team, however, is more in line with the management function, in that estimates of defensive expenditures provide a valuable indicator for policymakers in their own right.

The interpretation of any numbers that are classed as 'defensive' is a highly debated subject. It has been suggested that the information may provide an explanation for measured changes in productivity since abatement and other forms of defensive expenditures divert resources from conventionally measured production.[1] This argument, however, assumes that the observed abatement expenditures are representative of the true opportunity costs of environmental protection, which is questionable.

In order to begin to investigate defensive expenditures it is necessary to delimit the concept in workable terms. A good working definition is offered by Leipert (1989), who defines defensive expenditures as 'an indicator of the total monetary burden which society bears annually for the regulation of environmental degradation and damages induced by the economic use of the environment in the past and present periods. They also include future oriented expenditures.'

The scope and content of defensive expenditures is also defined well by Leipert (1989):

Preventive environmental protection activities

(a) changes in the characteristics of goods and services;
(b) changes in consumption patterns or production techniques;
(c) treatment or disposal of residuals in separate environmental protection facilities;
(d) recycling;
(e) prevention and degradation of landscape and ecosystems.

Environmental restoration (reactive environmental protection)

(a) reduction or neutralisation of residuals;
(b) changes in spatial distribution of residuals, support of environmental assimilation;
(c) restoration of ecosystems, landscape and so on.

Damage avoidance from environmental deterioration

(a) evasion activities (for example, permanent relocation);
(b) screening activities (for example, noise abatement windows).

Treatment of environmental damages

(a) repair of buildings, production facilities, historical monuments and so on;
(b) additional cleaning activities;
(c) other compensatory activities.

Environmental expenditures can be capital or current (operating) spending that is incurred because of, and which can be directly related to, the pursuit of an environmental objective. Hence, spending on processes that are environmentally benign but that also generate revenues that exceed expenditures are excluded since they are assumed to be premised on commercial decisions (for example, retrofitting with energy saving technologies). This delimitation can reasonably be applied to industry (where interest has been greatest and data are reasonable) and the public sector (where some studies are available). In the case of households, defining what constitutes defensive expenditures is particularly problematic and subjective and methodological guidelines are very sparse (see next section).

There are a number of methodological approaches for the collection of defensive expenditures and these have formed the basis for most work at the national statistical level. These will now be presented in turn, followed by a survey of previous and ongoing initiatives in the study countries.

7.2 OECD AND EUROSTAT APPROACHES

In the 1990s specific methodologies were developed by different international organisations. Both the OECD and the Statistical Office of the European Communities developed their own methods and classifications.

The OECD has developed separate methodologies for the estimation of both environmental expenditure and natural resource stocks. Since 1989, OECD has proposed a methodology for the calculation of Pollution Abatement and Control Expenditure (PAC). This reinforces the work which had been pursued since the late 1970s and aims at integrating environmental and economic decision-making more systematically and effectively. The methodology presented here is founded on the 1991 Recommendation from the OECD Council on Environmental Indicators and Information and the elaboration detailed in various monographs.[2]

Pollution abatement and control (PAC) activities are defined as purposeful activities aimed directly at the prevention, reduction and elimination of pollution or nuisances arising as a residual of production processes or consumption of goods and services.

This definition specifically excludes expenditure on natural resource management and activities such as the protection of endangered species, the establishment of natural parks and green belts, as well as activities to exploit natural resources (such as the supply of drinking water). As noted in the previous section, expenditures improving workplace protection or production processes for commercial and technical reasons are excluded even when they have environmental benefits.

Most OECD member countries, in their statistical approaches, only include expenditure that is directly aimed at environmental protection. This is often problematic, since it is difficult to distinguish expenditure motives, particularly in the case of integrated technology investments. In the absence of detailed survey data, many countries will employ a pragmatic solution, attributing a fixed proportion of expenditure to pollution abatement and control and the rest to ordinary expenditure. In the US, for example, 70 per cent of total expenditure on the collection and disposal of municipal waste is attributed to pollution and abatement control and 30 per cent is assumed to be ordinary expenditure.

According to the OECD approach the expenditure is broken down according to the type of expenditure (investment and current expenditure), the environmental domain and the sector incurring and/or financing the expenditure (public sector, private sector, households). The distinction between the entity executing the expenditure (abater principle) and the entity financing it (financing principle) avoids double counting when applying an integrated environmental accounting framework with all sectors of the economy.

In response to the EU demand for environmental data introduced with the Fourth and Fifth Environmental Action Programmes, Eurostat developed the SERIEE,[3] a common framework for the collection and presentation of economic data on the environment. It is a satellite accounting system, which refers to the domestic territory. Unlike the SEEA system, which will be described in the next section, it does not constitute a complete information system and does not integrate conventional accounts and environmental accounts to produce any estimates of 'Ecological Domestic Product'.

The approach is mainly monetary and like the OECD methodology it distinguishes between the execution and the financing of environmental expenditures. The main objectives of the system are:

- to present monetary flows associated with environmental protection;
- to determine the effects of environmental protection on the economy;
- to prepare the groundwork for defining indicators;
- to take the environment into account in the decision-making process of each economic unit and public bodies in particular.

The SERIEE framework examines those activities aimed at reducing and/or preventing pressure on the environment, monitoring and restoring the environment and exploiting the environment.

The general framework consist of two main accounts:

Environmental Protection Expenditure Account (EPEA) This highlights the execution and the financing of the environmental protection expenditure and accords with the concepts presented in Version II of the SEEA.

Natural Resource Use and Management Account This is primarily an 'economic' account which describes monetary flows related to the management and use of natural resources, such as the management of water, forest, soil, energy and so on, including recycling and recovery activities for the part not covered by the EPEA. As an example, energy and raw materials saving are included in this account.

For the purpose of our study we will focus on the EPEA, partly because the latter is still under development. The methodological approach taken for EPEA served as the groundwork for the revision of the UN system dealing with defensive expenditure (UN et al., forthcoming). However, the SERIEE approach designed in 1994 proved to be far too complex to be implemented. In addition, more recently, OECD and Eurostat have systematically co-operated with a view to harmonising their respective methodologies and at the same time avoiding the duplication of work. As a result, in 2002 they launched a common questionnaire on environmental protection expenditure and revenues using common definitions and classifications.

Like the system for Pollution Abatement and Control developed by the OECD, the EPEA records environmental protection expenditure associated with 'action or activities aimed at the prevention, reduction and elimination of pollution as well as any other degradation of the environment' (European Commission, 1994). Only those activities whose prime objective is environmental protection are included. These are called *characteristic activities*. Actions and activities which have a favourable impact on the environment while pursuing other goals are not taken into account.

Characteristic activities are determined on the basis of the Single European Standard Statistical Classification of Environmental Protection Activities (CEPA).[4] They are disaggregated in the following way:

Pollution media

(a) protection of ambient air and climate account;
(b) wastewater management account (excluding groundwater);
(c) waste management account;
(d) protection of soil and groundwater account;
(e) noise and vibration abatement account;
(f) protection of biodiversity and landscape account.

Type/module

(a) prevention and reduction of pollutant input;
(b) end-of-pipe techniques (filtering, transportation, storage);
(c) measurement, control and monitoring;
(d) administration and management.

7.2.1 Recent Developments

The approaches of the OECD and Eurostat are converging to a large extent. In the following the classifications used recently (for example, OECD, 2003) are presented. Some recent data on environmental expenditure will be presented in Chapter 15, according to the classifications illustrated below.

Recent data published by the OECD and produced jointly with Eurostat (OECD, 2003) focus on pollution abatement and control (PAC), as defined in the previous section. Basically, PAC expenditure comprises the flow of investment, internal current expenditure and fees that are directly aimed at pollution abatement and control, and which are incurred by the public sector, the business sector, private households and specialised producers of PAC services.

PAC expenditure together with expenditure related to the protection of biodiversity and landscapes (nature protection) form part of Environmental Protection (EP) expenditure. The scope of environmental protection is defined according to the Classification of Environmental Protection Activities (CEPA), which distinguishes nine different environmental domains, as illustrated previously. However, data are usually available for the domains that account for the largest share of expenditure: protection of ambient air and climate, wastewater management and waste management.

To guard against double counting, this approach distinguishes between the execution (*abater principle*) and the financing (*financing principle*) of PAC activities. These principles are illustrated in Table 7.1. Nevertheless, only a few OECD countries (for example, the Netherlands) evaluate expenditure according to both principles. Most countries only use and produce statistics according to the abater principle and do not include financial transfers. Thus, they only take into account where the expense originates.

PAC expenditure is reported according to the following breakdown:

- type of expenditure (investment, current expenditure, and so on);
- environmental media (air, water, waste, and so on);
- economic sector (public sector, business sector, specialised producers and households).

Type of expenditure

Investment expenditure is defined as outlays in one given year (purchase and own account production) for machinery, equipment and land used for abatement and control purposes (fixed assets). It comprises both end-of-pipe investments and investments in integrated technologies.

Table 7.1 Abater and financing principles

Public sector	Private sector
Investment expenditure + Current expenditure − Receipts from by-products of PAC activity = PAC expenditure according to the *abater principle* (*EXPENDITURE 1*)	Investment expenditure + Current expenditure − Receipts from by-products of PAC activity = PAC expenditure according to the *abater principle* (*EXPENDITURE 1*)
+ Subsidies to the private sector − Fees/charges from the private sector = PAC expenditure according to the *financing principle* (*EXPENDITURE 2*)	− Subsidies from the public sector + Fees/charges to the public sector = PAC expenditure according to the *financing principle* (*EXPENDITURE 2*)

Source: OECD (1996).

Current expenditure includes PAC expenditure for the use of energy, materials, maintenance and own personnel, including those for operating environmental protection equipment. General administration, education, information, research and development are also included.

Subsidies and transfers include all types of transfers financing PAC activities in other sectors, including transfers to and from other countries.

Fees and purchases include all purchases of PAC services, from both public and private producers. It corresponds to PAC activities done outside the enterprise and excludes fines and penalties.

Finally, *revenues* of the public service and specialised producers of services rendered together with *receipts from by-products* that are sold or used internally (e.g. energy generated or material recovered, as a result of waste treatment) are also to be considered.

It is to be noted that this methodology considers the expenditure and not the cost of pollution abatement and control. The latter can be calculated from the PAC expenditure as the sum of current expenditure and the ratio of investments for the year considered. In fact, the PAC cost reflects accurately the economic effects over time, but requires some assumptions about the number of years for which capital goods are used, on interest rates and so on.

Environmental media

The scope of these expenditures is defined according to the CEPA.

Waste includes municipal waste as well as industrial waste, which in turn includes hazardous waste, ordinary waste and inert or heavy waste (waste from the extractive industry and power stations, and demolition waste). It excludes water waste. PAC activities for waste comprise preventive measures to limit the amounts and harmful effects of waste generated from the final consumption of goods and to limit the production of industrial waste or lessen its harmful effects in terms of collection and transport; treatment and disposal; exploitation of waste; regulation and monitoring.

PAC activities for *water* comprise: collection and purification of waste water; combating of pollution in the marine environment; prevention, control and monitoring of surface water pollution; combating of pollution of inland surface waters; prevention and combating of thermal pollution of water; abatement of groundwater pollution; regulation and monitoring.

PAC activities for *air* comprise monitoring and regulation of atmospheric pollution and climate change, prevention of air pollution and climate change linked to the production process, installation of non-polluting technologies and elimination of emissions at source, as well as end-of-pipe technologies.

There is little data available for other PAC activities and therefore these are generally reported in an aggregated way under '*Others*'.

Economic sectors

The *public sector* comprises state, local governments and communities.

The *business sector* covers agriculture, hunting and fishing (ISIC/NACE 11 and 13), forestry (ISIC/NACE 12), mining and quarrying (ISIC/NACE 2), manufacturing (ISIC/NACE 3), electricity, gas and water (ISIC/NACE 4), construction (ISIC/NACE 5), transport, storage and communications (ISIC/NACE 7), other services (ISIC/NACE 6, 8 and 9 except government).

The *specialised producers* of environmental protection services include both publicly and privately owned enterprises. These are mainly activities within the ISIC/NACE 90 (collection and treatment of sewage, collection and treatment of solid waste sanitation, remediation and so on). However, expenditure for producing market environmental goods (for example, environmental protection equipment) are excluded.

Households' expenditure includes sewage treatment, and the purchase, operation and maintenance of air pollution control devices. This includes price differentials for unleaded gasoline or the costs of proper adjustment of engines.

7.3 THE SEEA APPROACH: TOWARDS AN INTERNATIONALLY HARMONISED FRAMEWORK

The Integrated System of Environmental and Economic Accounting, commonly referred to as SEEA, was initiated by the United Nations Statistical Office (UNSO), with the aim of constructing a monetary satellite system designed to be integral to the standard System of National Accounts (SNA). The methodological approach – described in a publication of the United Nations Statistical Office (1993) – was tested on an experimental basis in several countries: Colombia, Ghana, Indonesia, Japan, Mexico, Papua New Guinea, the Philippines, the Republic of Korea, Thailand and the USA. However, only parts of the SEEA were compiled in these tests because of lack of data as well as difficulties and uncertainties in applying the proposed valuation techniques. Moreover, several developed countries are now focusing on physical accounts. In this spirit, UNSO, in co-operation with experts and representatives of international organisations, revised the SEEA framework in order to integrate the twin concerns of providing both monetary and physical information on the environment and making this framework more operational. This revision is known as the SEEA review (UN et al., forthcoming).

The original version of the SEEA contained four parts:

Part A: Environment related disaggregation of conventional national accounts The basic framework for the SEEA is provided from the disaggregated form of SNA accounts, containing transactions and other economic flows and stocks of the SNA which are of special relevance to the measurement of the environmental impact of economic activities. As an example, the starting point for the natural asset accounts of the SEEA is the non-financial asset accounts of the SNA.

Part B: Physical data on environmental–economic interrelationships This part is a description of the interrelationships between the environment and the economy in physical terms. It contains physical data on the flow of natural resources, pollutants and residuals.

Part C: Valuation of the economic use of the environment The imputed cost of the use of natural assets is evaluated on the basis of (i) market valuation according to the SNA concepts of non-financial asset accounts; (ii) maintenance evaluation, which estimates the cost necessary to maintain the present level of natural assets; (iii) contingent valuation, which could be applied especially for consumptive services of the natural environment.

Part D: Extension of the production boundary of the SNA Extensions have been applied especially for household activities in order to better understand the social and demographic aspects behind the impact of human activities on the environment and the repercussions of a deterioration in the environment on human welfare (extension of production boundary with regard to household activities). In addition, environmental services produced by nature (for disposal purposes, productive purposes such as agriculture and consumption purposes (physiological, recreational and so on)) are taken into account (introduction of environmental services as an output of productive activities of the natural environment). Furthermore, the possibility of treating both internal and external environmental services as production activities is introduced, thereby attributing a value to the output (externalisation of internal environmental protection activities).

For a more detailed discussion of the overall SEEA framework and issues relating to it, the reader is referred to UN (1993) or Lutz (1993). In the scope of our analysis on environmental expenditure, Version II is the most relevant and will be considered in more detail.

The objectives of SEEA were twofold:

(i) to determine the *actual cost* incurred to prevent or to restore the negative impacts of economic activities on the natural environment and to prevent or compensate for further or indirect negative impacts of the repercussions of a deteriorated natural environment; and
(ii) to determine the *non-financial assets* comprising opening stocks, price and volume changes during the accounting period and closing stocks.

To achieve these objectives the input–output framework (supply and use table) and the non-financial asset accounts were disaggregated to identify:

- environmental protection activities (prevention and restoration), as well as activities aimed at avoiding or treating damages due to the deterioration of the environment;
- consumption or use by units of goods and services which contribute to the protection or the restoration of the environment or compensation for nuisance caused by its deterioration;
- fixed asset formation used for environmental protection or to compensate deterioration effects;
- non-financial assets according to different categories (assets produced by man or by nature, non-produced assets).

Having determined all environmental protection activities and the consumption or use by units of goods and services which contribute to the protection or restoration of the environment, this version of SEEA proceeded to compute the fixed asset formation and the description of non-financial assets according to different categories. Finally, it provided accounting rules for their changes in volume. However, this second objective is not relevant for the purposes of this chapter on defensive expenditure.

Although the revised SEEA handbook takes further many of the ideas outlined in the original version, a substantial restructuring has been carried out during the SEEA revision by the so-called London Group, a group of representatives of statistical offices and international organisations which was set up in 1993 to provide an informed forum of statisticians and practitioners to develop and implement environmental satellite accounts linked to economic accounts.

We take this opportunity to provide a synthetic overview of major changes and then highlight what part of the SERIEE has been adopted for the purpose of SEEA. The new version, instead of proposing a very rigid scheme for supply and demand tables, requiring extensive work in valuation in order to adjust for the cost of depletion of natural resources and degradation of environmental media, as was the case for the 1993 SEEA, provides a framework of different accounts, from which every country can choose the most appropriate ones.

Like the 1993 SEEA, the revised SEEA includes tables in purely physical terms, showing the links between the use of goods manufactured in the economy, natural resources and residual output to the environment (that is, pollution). The revised SEEA, however, tries to highlight the policy implications by developing further this type of analysis and calculating the 'total material requirement' to show the dependence of the economy on a volume of material inputs which typically expand as the economy develops. More importantly, the revised SEEA introduces the idea of 'hybrid accounts', which show the conventional national accounts, consistent with SNA, augmented by physical data for residual outputs and resource inputs. We refer here to the NAMEA type of accounts that were originally developed by the Dutch Statistical Office. The valuation methods highlighted in the 1993 SEEA for the development of monetary accounts to be fed into the demand and supply tables are still subject to controversy. However, the new handbook includes descriptions of all techniques setting out the arguments for and against their use. No consensus has yet been reached on how to value resources, their depletion and degradation and whether it is legitimate to introduce these values into conventional statistics. An entire chapter of the revised SEEA is devoted to environmental expenditure, which has certainly gained interest and policy relevance in comparison to the previous version.

The SEEA approach certainly draws very much on the Eurostat and OECD experience. In the following chapter we shall show which approach to environmental expenditures has been taken for the SEEA, as illustrated in Chapter 5 of SEEA on 'Accounting for economic activities and products related to the environment', where the expenditures connected both to the protection of the environment and the management and exploitation of natural resources are addressed. However, it is recognised that there have been advances in methodology and in practical experience in the area of protection of the environment.

Environmental Protection Expenditure Accounts are based to a very large extent on the Eurostat and OECD approach and reference is made to the same classifications, for example, the CEPA classification for environmental protection activity and the distinction between current expenditure and investment. As with the SERIEE system, the SEEA presents a series of supply and use tables, which clearly distinguish between execution and financing of activities and use/consumption and production:

- supply and use tables for environmental protection goods and services;
- national expenditure on environmental protection by components and by users/beneficiaries (see Table 7.2);
- financing of national expenditure for environmental protection.

The supply and use table is calculated on the basis of the use of the input required by various actors (specialised producers, non-specialised producers, public sector, households, rest of the world). The total uses (capital expenditure and current expenditure) at the national level, including financing by the rest of the world, measure the *National Expenditure for Environmental Protection*, which is calculated as follows:

final and intermediate consumption of environmental protection products by resident units, other than those of the environmental protection producers

+ consumption of specific products: final and intermediate consumption of characteristic services, connected and adapted products
+ gross capital formation for characteristic activities
+ gross capital formation for specific products (connected, adapted products and characteristic services)
+ specific transfers: subsidies for characteristic services, connected or adapted products, other current and capital transfers
− financing of current or capital use by the rest of the world.

Table 7.2 National expenditure by components and by users/beneficiaries

Components of national expenditure for environmental protection	USERS/BENEFICIARIES								
	Producers				General government as collective consumer		Households as actual consumers	Rest of the world	Total
	Specialised producers		Other producers (industry)						
	General Government (GG) & Non-Profit Institutions Serving Households (NPISH)	Other	Non-specialised	Non-characteristic	Central Government (CG)	Local Government (LG)			
1 Consumption of specific products									
1.1 Final consumption of characteristic services									
market									
non-market									
1.2 Intermediate consumption of characteristic services									
market									
ancillary									

Table 7.2 (continued)

Components of national expenditure for environmental protection	USERS/BENEFICIARIES								
	Producers				General government as collective consumer		Households as actual consumers	Rest of the world	Total
	Specialised producers		Other producers (industry)						
	General Government (GG) & Non-Profit Institutions Serving Households (NPISH)	Other	Non-specialised	Non-characteristic	Central Government (CG)	Local government (LG)			
1.3 Final consumption of connected products adapted products									
1.4 Intermediate consumption of connected products adapted products									
2 Gross capital formation in (1) for characteristic activities									

3	Gross capital formation
	in specific products
	in connected products
	in adapted products
	in characteristic services
4	Specific transfers (not counterpart of item 1, 2, 3)
	4.1 Subsidies on production
	characteristic services
	connected products
	adapted products
	4.2 Other specific transfers
	current
	capital
5	Total uses of residents' units (1 + 2 + 3 + 4)
	current
	capital
6	Financed by the rest of the world
	current uses
	capital uses
7	National expenditure for environmental protection (5–6)
	current
	capital

7.4 CONCLUSIONS

There are essentially two fundamental issues to be considered in evaluating the strengths and weaknesses of different methodologies. The first is technical, namely how accurately the classifications used are able to isolate those expenditures which can really be considered 'defensive'. This has to be balanced with the second issue of how practicable it is to obtain the relevant data and the ability to interpret the numbers correctly.

The work undertaken in this research project, and hence in this book, on environmental expenditure has not involved development of a further methodological approach. Rather, we report in Chapter 15 on what statistical evidence existed in the four case study countries at the time when the project was carried out, and also report more recent data, showing some improvement in the data availability and harmonisation across countries. We have gathered data from official and unofficial survey sources and present this information broken down by sector. Where possible data are also presented by environmental theme or media – according to the approach adopted in the individual country (see Chapter 15).

A principal objective of this book is to identify the gaps and inconsistencies that exist in current reporting procedures compared with the requirements implicit in the accounting frameworks developed. In particular, we attempt to highlight key discrepancies between the SERIEE methodology and the existing national estimates, whilst identifying priority areas for future research efforts on methodological approaches. Our conclusions broadly coincide with the areas of revision undertaken by the London Group on Environmental Accounting and now endorsed by the United Nations, European Commission, OECD, IMF and World Bank.

It is noteworthy that there has been a substantial effort over time to harmonise methodologies at international level. Today the OECD and Eurostat use the same framework for pollution abatement and control expenditure and are also jointly working to improve data availability on biodiversity and landscape, in order to have a more comprehensive picture of overall environmental expenditure. The incorporation of the SERIEE approach into the SEEA system also represents a step towards more comprehensive and harmonised accounts.

NOTES

1. See Peskin (1999), who cites Denison (1979) and Jorgenson and Wilcoxen (1990).
2. The methodological concepts were laid down in OECD (1996). The most recent monograph both on methodological issues and data is OECD (2003).

3. SERIEE is the French acronym for the European System for the Collection of Economic Information on the Environment.
4. The CEPA was prepared jointly by UNECE and Eurostat in 1994 and revised in 2000.

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PART II

Results

INTRODUCTION

In Chapters 8 and 9 comprehensive results of estimates are presented of damage costs caused by air pollution using monitored concentration data and modelled emissions respectively. These damages were calculated using the EcoSense model developed at the Institut für Energiewirtschaft und Rationelle Energieanwendung (IER), which gives estimates of the impacts on main receptors: human health, crops and materials. In addition, in Chapter 9 damages caused by air pollution are attributed to countries and economic sectors of origin. In these chapters results are presented by receptor, as is the case in Chapter 10, which consists of a review of existing studies on the evaluation of damages to crops. In Chapters 11 to 13 results are presented by media, Chapter 11 dealing with damages to forests and ecosystems, Chapter 12 with damages to water and Chapter 13 with damages to land. Finally, Chapter 14 presents estimates for global warming damages and Chapter 15 provides a detailed analysis of defensive expenditures.

8. Estimates of damage costs from air pollution to human health, crops and materials

Bert Droste-Franke, Wolfram Krewitt, Rainer Friedrich, Alfred Trukenmüller, Marialuisa Tamborra, Gianluca Crapanzano, Marcella Pavan, Paul Watkiss, Mike Holland, Katie King, Anil Markandya, Ian Milborrow, Alistair Hunt, Onno Kuik, Kees Dorland, Frank A. Spaninks and John F.M. Helming

8.1 INTRODUCTION

The current chapter deals with the estimation of impacts caused by anthropogenic emissions based on observed air quality data. The core study contains damage estimations for the countries of Germany, Italy, the Netherlands and the United Kingdom using the recommended exposure–response functions (see Chapters 4 and 5). First, the air quality data and methodology applied for each country are described in detail. Then the natural background concentrations used for the attribution of damages to anthropogenic origin are discussed. Afterwards, the results are presented in detail for each country and impact category (mortality, morbidity, crops and materials) and summarised by valuing the physical impacts in monetary terms. A sensitivity analysis follows which includes, for the UK only, the assessment of damages in different years, damages caused by further pollutants for which no assessment is carried out in the core analysis, and damages estimated by using alternative exposure–response functions for some human health effects. In the final part of this section conclusions are derived from the results.

8.2 DETAILED METHODOLOGY

The availability of measurement data for air pollutants differs much between Germany, Italy, the Netherlands and the UK. While concentration maps of geographical high resolution are already available in some countries, in others they were derived by interpolating measured concentration data. In the following the available data sets and interpolation methods, as well as the stock at risk data applied, are described in detail for each country of study.

8.2.1 Air Quality Data

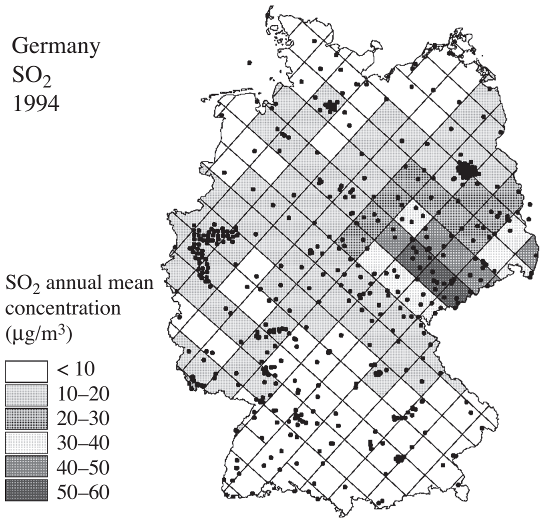
This chapter considers the impact pathway of the following air pollutants: sulphur dioxide (SO_2), nitrogen dioxide (NO_2), total suspended particles or particulate matter with a diameter smaller than 10 μm (TSP or PM_{10}) and ozone (O_3). When available, other pollutants are taken into account, such as carbon oxide (CO) and some heavy metals.

Germany

Interpolation of measured concentration data
For the estimation of damages caused by measured pollution data, concentration maps were created for Germany by interpolating data reported by NILU and the German Environment Agency. The statistical method of ordinary kriging (*sic*) was carried out using the Geographic Information System (GIS) ARCINFO version 7.11 (Burrough, 1993; Heinrich, 1992). The measurement data were interpolated to a 10×10 km^2 grid before they were averaged over 25 grid cells and distributed to the 50×50 km^2 – EMEP (EMEP50) grid cells. In the following the available measurement data and estimated data maps are individually described for each pollutant.

Sulphur dioxide (SO_2) was measured at 515 stations all over Germany in 1994 (Bräuniger, 1997) providing a spatial density of observed data that was high enough to get a reliable concentration map through data interpolation to the EMEP50 grid. The estimated SO_2 map is shown in Figure 8.1.

The highest concentration of SO_2 in 1994 was observed in the eastern part of Germany due to high emissions of its power plants. From 1989 to 1994, SO_2 emissions fell by 40 per cent, but they still remain the most important pollutant of Germany. In 1994, the annual mean concentration values did not exceed the health-based standard for SO_2 of 140 μg/m^3 which has been set by the German government (TA Luft, 1986). The last exceedance of this level was observed in 1992 in some regions in the eastern



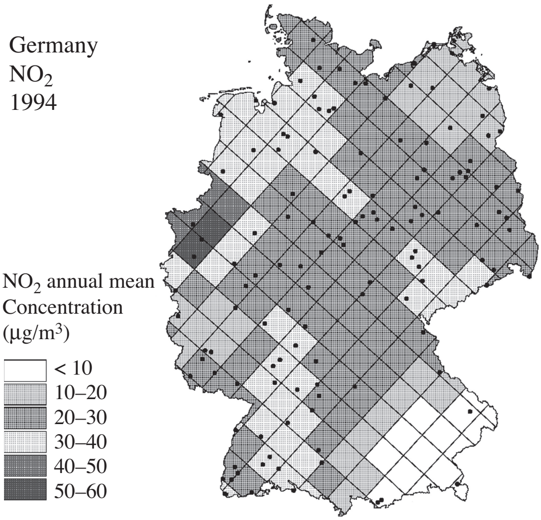
Source: German Federal Environment Agency, Berlin.

Figure 8.1 SO_2 annual mean concentrations and measurement stations (marked as points) in Germany, 1994

part of Germany (UBA, 1997). However, the critical levels which are reported for forests and natural vegetation (20 μg/m^3) and agricultural crops (30 μg/m^3) (UBA, 1996) were exceeded in large areas of eastern Germany in 1994.

In order to avoid influences of road traffic, only data of stations suitably distant from roads have been used for the interpolation of *nitrogen dioxide (NO_2)*. These were identified from the characterisation of measurement stations defined by the German Environmental Agency (Bräuniger, 1997). Although the remaining 211 measurement stations from all over Germany provided a density of measurements that was quite sparse in some regions, a kriging (*sic*) interpolation was possible. The concentration map is shown in Figure 8.2.

The highest concentration of NO_2 was observed in regions with high traffic density, especially in the industrial centres of western Germany. The interpolated concentration values for 1994 did not exceed 60 μg/m^3 in Germany, which is lower than the human health standard of 80 μg/m^3 reported by the German government (TA Luft, 1986). The monitored concentrations levels in 1994 for NO_2 were found to be higher than the critical level for vegetation of μg/m^3 (UBA, 1996) in large areas of the western part of Germany.

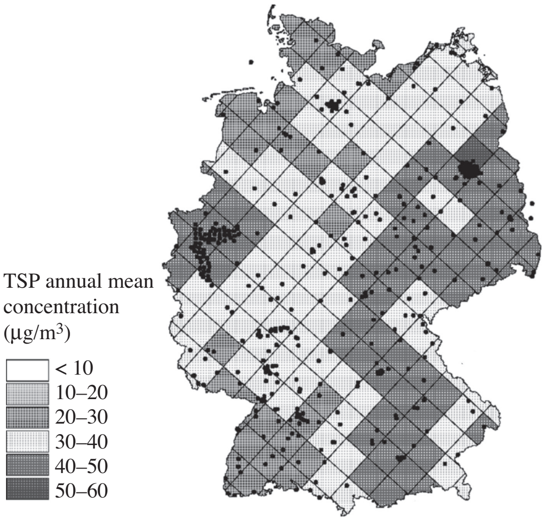


Source: German Federal Environmental Agency, Berlin.

Figure 8.2 NO_2 annual mean concentrations and measurement stations (marked as points) in Germany, 1994

Data from 434 measurement stations were used for the interpolation of *Total Suspended Particulates (TSP)* concentration (Bräuniger, 1997). In urban areas of Germany about 85 per cent of the measured TSP consists of particulate matter with a diameter smaller than 10 µm (PM_{10}) (VDI, 1990), which is especially implicated with impacts on human health. In rural areas this fraction is lower and may decrease under special conditions to about 10 per cent of TSP. Figure 8.3 shows that the TSP annual mean concentration was between 10 and 60 $\mu g/m^3$ in 1994. The observed concentrations are thus much lower than the health related standard of 150 $\mu g/m^3$ which the German government reported for TSP (TA Luft, 1986). In regions where industrial production is low (for example, by the coast) and therefore the main fraction is of natural origin (for example, suspension of dust and sea salt), the measured annual mean concentrations were between 20 $\mu g/m^3$ and 30 $\mu g/m^3$.

Two different data sources were used for the interpolation of *ozone (O_3)* concentration data. The interpolation of six-hour annual mean concentration (annual average between 12 noon and 6 p.m.) and AOT60[1] values was carried out using data reported from the German Environmental Agency while AOT40[2] data were taken from NILU (Dauert, 1997; Hjellbrekke, 1996, 1997). According to the current convention in Europe, the AOT40



Source: German Federal Environmental Agency, Berlin.

Figure 8.3 TSP annual mean concentrations and measurement stations (marked as points) in Germany, 1994

values for forests and agricultural crops were accumulated during daylight hours when the mean global radiation exceeded 50 W/m^2. Investigations showed that averaging over the period of 6 a.m. to 6 p.m. results in an average deviation of only about 3 per cent compared to estimations in which detailed information about the radiation were applied (Köble et al., 1997; Posch et al., 1997). Therefore, this time period was assumed to be daylight hours in the current analysis.

Measurements influenced by traffic emissions reflect only the local situation concerning the O_3 concentration, because road traffic emits large amounts of NO which reduces O_3 and thereby decreases the concentration locally. Therefore, only measurements of stations not directly influenced by road traffic were considered for the calculation of the concentration maps which cover the whole area of Germany. As O_3 concentrations show additional variations with altitude, a classification method was applied for the selection of measurement stations which considers the classes defined by the German Federal Environmental Agency (UBA), the ratio of the measured ratio NO/NO_2 as well as the altitude (Köble, 1998; Köble et al., 1997). This method yielded 130 measurement stations in Germany that could be considered for the assessment. No further classification was applied for the European measurement sites of EMEP/NILU, because they are all restricted to rural areas and thereby not influenced by road traffic.

The World Health Organisation (WHO) reports a running eight-hour average of >60 ppbVh as the threshold for impacts on human health (although some recent epidemiology studies point out that no threshold for ozone effects exists). Following this definition, the AOT60 value gives a first indication of the impacts on human health; no health effects would be expected on the assumption of a 60 ppbV threshold if the AOT60 value was zero. Figure 8.4 shows AOT60 > zero in all grid squares in Germany in 1994. Therefore, the AOT60 map can only be used for qualitative statements about the exposure of humans to ozone. The highest potential of health damages is expected in the south-west and some regions of the central part of Germany. The exposure–response functions for human health effects recommended for the present study require six-hour annual mean concentrations of O_3 and thereby consider no thresholds. It can be seen that the highest six-hour annual mean concentrations in 1994 (35 to 37.5 ppbV) were observed in the south and some parts of the centre of Germany in a pattern similar to the AOT60 map.

AOT40 over the growing season is used as an indicator of damages to crops and forests. Figure 8.4 shows that the interpolated AOT40 values are especially high in the south and the centre of Germany. The lowest values are observed by the coast. The critical level for crops of 3000 ppbVh was exceeded in all regions of Germany. For forests, the only region in which the critical AOT40 of 10 000 ppbVh was not exceeded was the coast where forestry is limited (see also Köble et al., 1997).

Only 22 stations within the monitoring programme of EMEP reported data on annual mean concentrations of *heavy metals* in aerosols for 1994 (Berg et al., 1996). Not all substances have been measured at all stations so that the only metals for which an order of magnitude of the concentration level could be derived in Germany for 1994 were cadmium (20 stations throughout Europe) and lead (22 stations throughout Europe).

The rough interpolation results in concentrations of about 0.25 to 0.30 ng/m^3 for cadmium and 10 to 30 ng/m^3 for lead in Germany (see also Berg et al., 1996). These are much lower than the critical levels reported for human exposure of 40 ng/m^3 and 2000 ng/m^3 respectively (TA Luft, 1986).

Italy

The Italian monitoring system

The Italian situation is quite complex, for several reasons. First of all, the Italian monitoring system is not very well structured and managed. In fact, a recent monitoring networks survey reports 645 stations over the Italian territory (with one monitoring station every 470 km^2 on average):

- 423 measuring SO_2 concentrations,
- 402 measuring NO_x concentrations,
- 369 measuring TSP concentrations,
- 153 measuring O_3 concentrations,
- 271 measuring CO concentrations,
- 86 measuring NMHC (non-methane concentrations), and
- 47 measuring THC (total hydrocarbons).

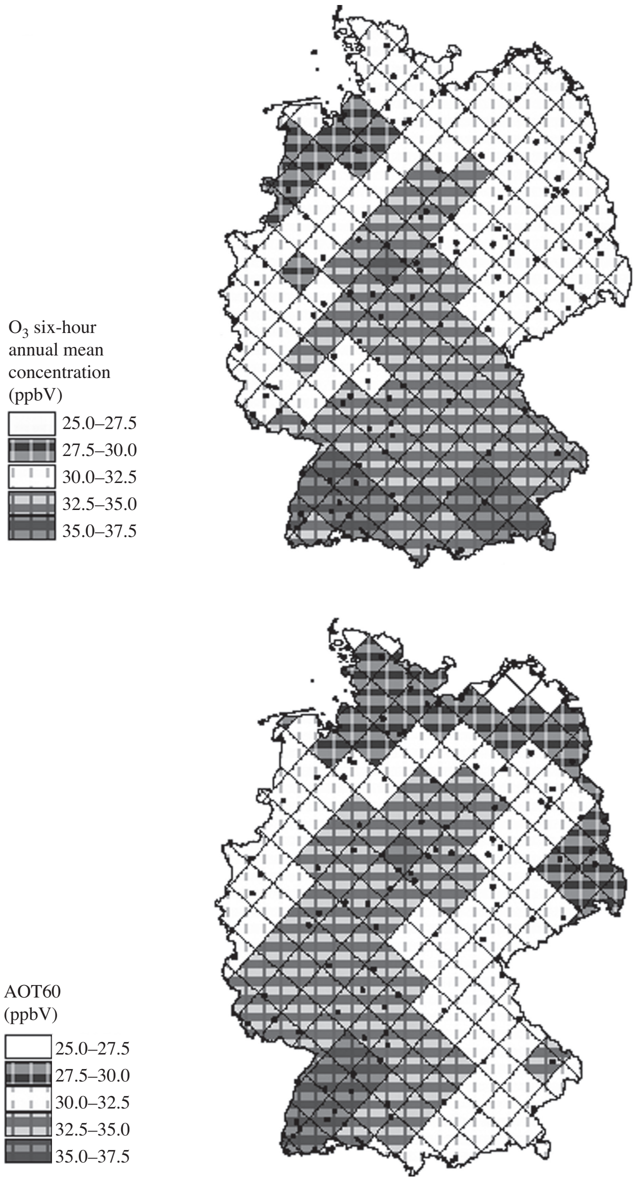
A more detailed analysis shows that the spatial distribution is not uniform: the existing stations cover only 60 of the 95 Italian provinces (corresponding to 60 per cent of the total area, 72 per cent of the population), and only 38 of them (31 per cent of the area, 55 per cent of the population) have more than one station on average every 500 km^2, while 15 out of the 60 provincial areas (19 per cent of the area, 12 per cent of the population) have less than one station per 1000 km^2. The Italian situation is peculiar because of a complex geography and meteorology and an uneven distribution of industrial activity (mainly in the plains). The presence of mountains adds uncertainty when interpolating data. The interpolation between the existing measurement stations is thus very difficult and results have quite low confidence.

The situation is alarming once we consider spatial disaggregation among northern, central and southern Italy, and among the different pollutants separately, as illustrated in Table 8.1 where the spatial coverage is reported.

As one can see from the table, the situation is quite good in the north and centre (area covered around 65 per cent in the north, 50 to 60 per cent in the centre, depending on pollutants), but even here only about 30 per cent of the area can be considered well covered. In the southern part of Italy the situation is much worse, with about 20 per cent of the territory covered for SO_2 and TSP measurements, and less than 10 per cent for other pollutants. In terms of population covered, the gap between the north–centre and the south would be even bigger.

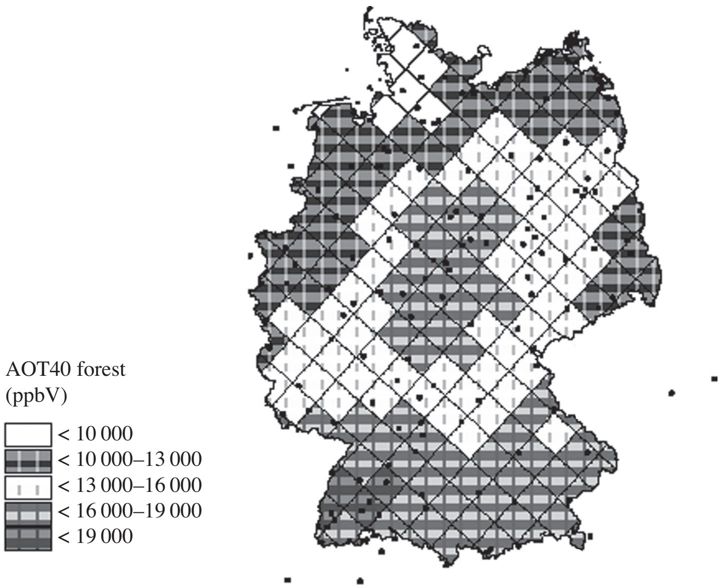
Measured concentration data

Few of the total existing measuring stations are working correctly, so that some stations cannot be considered for mean annual concentrations, because of the small number of observations (according to the law, measured mean annual concentrations are validated if they are calculated using at least 75 per cent of total possible values, that is, 6570 hourly concentrations



Source: Figures based on 1994 data by the German Federal Environment Agency (UBA, 1997) and NILU (Norsk institutt for luftforskening), Norway.

Figure 8.4 O_3 annual mean concentrations, AOT60 values, AOT40 values and measurement stations (marked as points) in Germany, 1994


AOT40 forest
(ppbV)
< 10 000
< 10 000–13 000
< 13 000–16 000
< 16 000–19 000
< 19 000

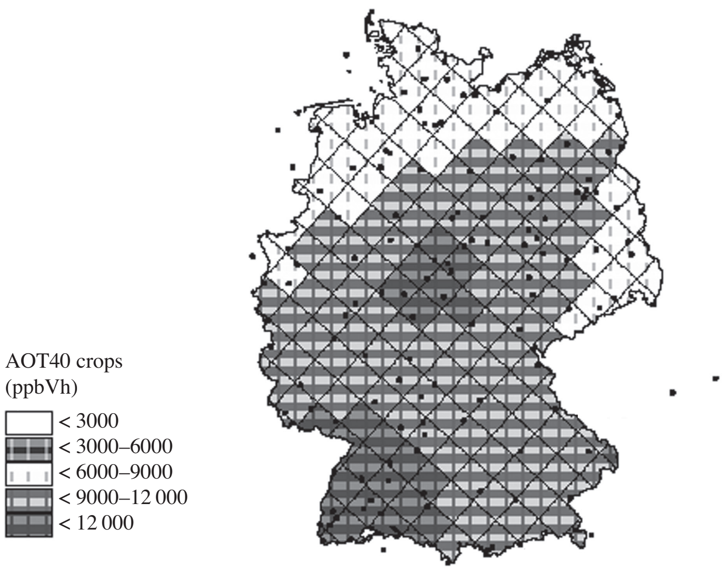

AOT40 crops
(ppbVh)
< 3000
< 3000–6000
< 6000–9000
< 9000–12 000
< 12 000

Table 8.1 Italian monitoring stations survey results – spatial coverage

	All stations		SO_2	NO_x	TSP	O_3	CO	NMHC	THC
	km²	%	%	%	%	%	%	%	%
Total	301 277								
Covered area	180 155	60	46	41	44	29	39	21	7
Good coverage[1]	94 788	31	21	24	19	7	13	4	3
Bad coverage[2]	56 844	19	13	9	14	12	14	13	3
North	97 763								
Covered area	78 432	80	65	65	65	53	63	36	13
Good coverage[1]	45 479	47	32	38	30	12	23	5	4
Bad coverage[2]	18 638	19	14	14	14	24	17	21	5
Centre	80 468								
Covered area	61 272	76	60	60	54	29	52	22	11
Good coverage[1]	40 944	51	29	36	26	5	21	7	5
Bad coverage[2]	13 640	17	17	17	21	9	21	16	3
South and islands	123 046								
Covered area	40 452	33	22	10	22	10	10	8	0
Good coverage[1]	8 365	7	6	4	6	4	0	2	0
Bad coverage[2]	24 567	20	10	0	10	6	6	6	0

Notes:
[1] More than one station per 500 km².
[2] Less than one station per 1000 km².

per year). Moreover, data are not always delivered to the central organisation responsible for collecting and elaborating all measurements.

Data from the Ministry of the Environment have been integrated with data from ISTAT (ISTAT, 1996, Statistiche ambientali), and they all refer to the year 1994. Data for the following pollutants were available: SO_2 (annual 50th percentile), NO_x (98th percentile), TSP (annual average), and O_3 (50th percentile). Measurements taken in stations falling in the same municipality have been averaged together, because they are too close to justify a spatial disaggregation.

The data set used for this study was obtained as follows: 237 measurements for SO_2 (in 120 different municipalities, so that they have been averaged in order to obtain only 120 different measurements), 145 for TSP (in 77 municipalities), 79 for NO_x (44 municipalities) and 29 for O_3 (20 municipalities). The possibility of obtaining some information for other pollutants was investigated, but not enough data were found for even an approximate interpolation.

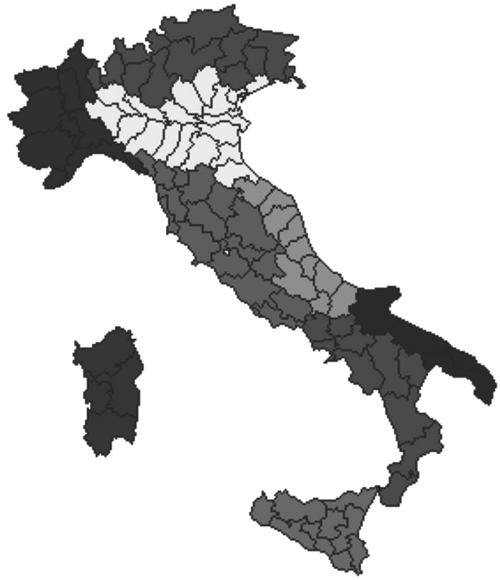
Nevertheless, data interpolation is considered in theory a better solution than air dispersion modelling, because of the particular topographic and meteorological situation of Italy. In fact, the modelling of mean annual concentrations can be done with dispersion models with reasonable computing time only if a coarse grid is used, and this would not account for the required spatial detail.

For all these reasons, we tried to develop a methodology for interpolating existing measurements, accounting for territorial specificity of different country areas, in order to calculate mean annual concentrations on a provincial basis (a finer resolution would not be feasible with the available data). Due to limited data availability, the application of this methodology was quite difficult, but the framework can be successfully applied as soon as more data are available.

In order to account for differences in Italian zones, the Italian territory was divided into nine sectors, according to the topography. The results of this repartition are displayed in Figure 8.5.

For some sub-regions, especially in the south of Italy, it was impossible to calculate average concentrations, because of the small number of measuring stations (in some cases, no stations at all occurred in the area). Four alternatives were taken into account:

1. to assign to the whole sub-region a constant value obtained by averaging all available measurements for the considered pollutant;
2. to remove the constraint of the division into sub-regions, in order to allow the extrapolation of measurements to sub-regions where no station was present;



Note: Italy has been divided into groups of regions that are represented using different shades. These groups have no political or geographical meaning; the choice has been made on the basis of data availability.

Figure 8.5 Homogeneous sectors division

3. to look for a statistical relationship between the available measurements and some demographic parameters, such as the population or the number of people working in particular sectors, in order to extrapolate existing data to places where no measurement is available;
4. to use modelling results, even if they do not account for spatial variability.

None of these options is considered much more reliable than the others. The first was implemented simply because it is the most straightforward. Results should be compared with modelled simulations when available. The second and third options were rejected for being too complex without clear additional benefit, and for a lack of reliable inputs, respectively.

For ozone concentrations, due to the low number of available measurements (data for 21 municipalities, seven of which were concentrated in the Emilia Romagna region, six in the Trento province, five in the central part of Italy, none in the south), no interpolation was made. No methodology was considered more meaningful than a rough averaging of the existing measurements across the nation. Therefore one single concentration value, that is, the average, was used for the calculation of health damage caused by ozone concentrations. No calculation was made for ozone damage to crops, since no information was available for calculation of AOT40 values.

Due to the scarcity of available measurements, the possibility of increasing the number of data using statistical techniques has been taken into account. For each pollutant for which data were available, we investigated the possibility of finding a correlation between the measured concentrations and some demographic parameter, such as the population or the number of employed people in different activity sectors (for example, industrial or commercial).

At the beginning the relationships were considered between the concentrations calculated in provinces with a high monitoring stations density and the mean annual emission rates in different activity sectors, such as traffic, industry or public heating, taken from the CORINAIR emissions inventory. This was rejected because it could not be applied to ozone and particulates and results for SO_2 and NO_X were unsatisfactory. In considering the correlation between measured concentrations and demographic data, the municipality scale was taken into account, identifying the relationship between the concentration data averaged over the municipality territory (the average area of municipalities is very small, about 37 km^2) and the population data (no estimation has been made using occupational data, since they are strongly correlated with the population).

Results show very weak correlation between all parameters investigated. It was therefore concluded that the use of average concentration values for each pollutant would not lead to worse results than extrapolation of concentrations according to demographic parameters.

The statistical analysis showed that samples are highly dispersed, so that an interpolation line cannot correctly describe the situation. The mean square deviations of the samples are usually of the same order of magnitude as the mean value, so that the error made by using the latter rather than extrapolated values is really small (with differences of 1 to 3 per cent in mean square error values of the estimates).

Results of interpolation of measured data

Concentration maps were obtained for SO_2, NO_x and TSP as reported in Figures 8.6 to 8.8.

For SO_2 and TSP, for which quite a high number of measurements are available, there is a region which is not covered (the Adriatic coastal region). For NO_x the number of regions which are not covered by measurements is higher still (mostly in the southern part of Italy).

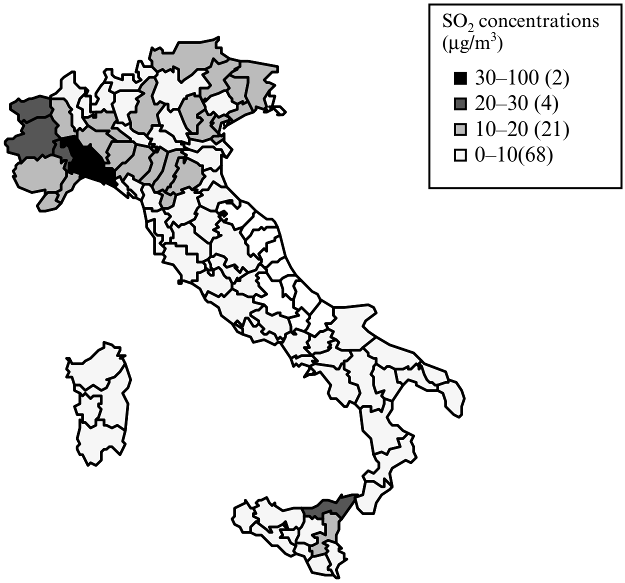


Figure 8.6 *SO_2 concentration map*

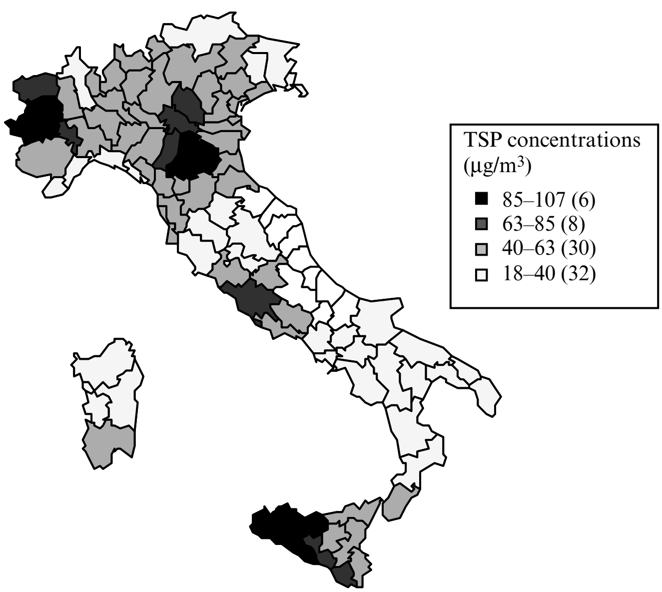


Figure 8.7 *TSP concentration map*

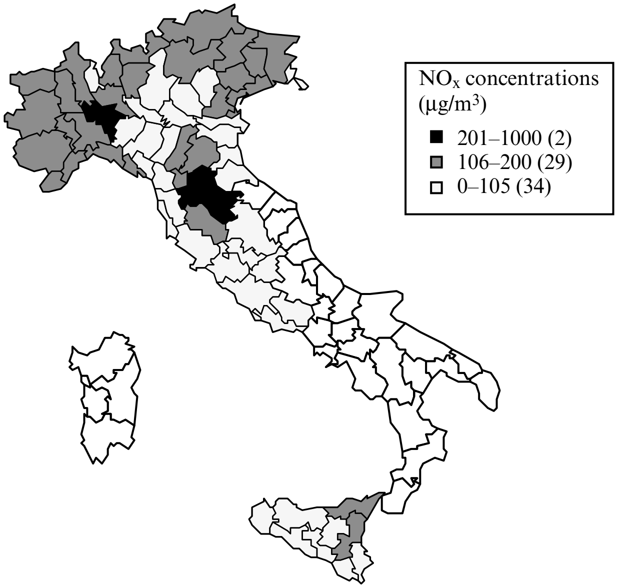


Figure 8.8 *NO_x concentration map*

A concentration map for PM_{10} can be derived from TSP by applying a factor of 0.85; the conversion from NO_x 98th percentile values to annual average values has been made by dividing by a factor of 2.6.

The Netherlands

Introduction

The scope of the Dutch assessment of impacts (except for impacts on forests and ecosystems) is presented in Table 8.2. The study benefited from the methodology that was developed under the ExternE Project (see European Commission, 1998). The assessment is based on exposure-response functions and the methodology presented in the first part of this book.

Measured concentration data

Air quality in the Netherlands is monitored by the National Institute of Public Health and Environmental Protection (RIVM) using the National Air Quality Monitoring Network (LML, in Dutch: *Landelijk Meetnet Luchtkwaliteit*). The annual average, 50, 95 and 98 percentile, 24-hour maximum and one-hour maximum pollutant concentration data for the LML stations are published annually in two publications. The 'Annual Air

Table 8.2 Pollutants and impact categories addressed in the Dutch assessment

	SO_2	NO_x	PM_{10}	O_3	CO	Heavy metals[a]
Health	X[b]	X[b]	X	X	X	X
Materials	X	X[c]		X		
Crops	X	X[d]		X[e]		

Notes:

a In the methodology section human health unit risk factors for As, Cd, Cr, Ni, Hg and Pb are given. For this Dutch case study no data on ambient concentration of Cr, Ni and Hg were readily available. Therefore no impacts could be estimated for these metals.

b SO_2 and NO_X emissions lead to formation of aerosols in the atmosphere. The contribution of these aerosols in the atmospheric dust concentrations is included in the PM_{10} category and impacts and damages are therefore not attributed to SO_2 and NO_x in the impact and damage assessment. SO_2 also leads to direct impacts on health and therefore this category is also shown in the impact and damage estimates. NO_x does not lead to direct health impacts and is therefore absent from the impact and damage estimate categories.

c Through acid deposition only.

d Impacts of NO_x on crops are indirect. They are due to ozone formation in the atmosphere. This impact category is therefore included in the ozone impacts on crops.

e Impacts from ozone on crops are related to the number of hours when the atmospheric ozone concentrations exceed 40 ppbV and when radiative forcing is higher than 50 W/m^2 and when the sky is clear (AOT40c).

Quality Survey' provides a broad overview of the air quality and the concentrations of a number of pollutants. The publication 'National Air Quality Monitoring Network: Results of the Measurements' (in Dutch: *Landelijk Meetnet Luchtkwaliteit: Meetresultaten*) gives more detailed data for the individual stations. This latter publication consists of four volumes: three for the regional stations and one for the city background and street stations.

The regional stations are considered representative for the air quality in a range of 5–50 km around the station, and are used by RIVM to calculate the overall distribution of air pollutant concentrations by means of interpolation. The city background stations are located in cities with 40 000–800 000 inhabitants and are considered representative of the air quality in comparable cities. Finally, street stations are located in urban streets with a high traffic intensity (>10 000 motorised vehicles per day) and are considered representative of the air quality in comparable streets (estimated at about 100 streets in the Netherlands).

Particulate matter is also measured outside the LML by organisations other than the RIVM. The 1994 measurement results of these stations,

Table 8.3 Air pollutants measuring stations in the LML and number of stations for 1994

		Number of stations			
Pollutant	Unit	Total	Regional	City	Street
SO_2	μg/m³	36	26	5	5
NO_x	ppbV	45	26	6	13
Particles: PM_{10}	μg/m³	18	9	4	5
O_3 (six-hour and annual average)	μg/m³	38	26	4	8
AOT40c	ppbVh	26	26	0	0
Oxidant	ppb	38	26	4	8
CO	mg/m³	21	5	4	12
Heavy metals: As, Cd, Pb, Zn	ng/m³	4	3	0	1
Heavy metals: Cr, Ni, Hg	–	0	0	0	0

Sources: Airbase (1998), RIVM (1996).

which are mainly located in urban and industrial areas, are also included in the Annual Air Quality Survey. Hourly average concentration data of SO_2, NO_x, PM_{10}, O_3 for a number of the LML stations are available on the website of the European Topic Centre on Air Quality through Airbase.[3]

Spatial distribution of air pollutant concentrations

In most exposure–response functions used in this study annual average concentration data are used. Only in the case of health damages due to ozone are annual average six-hour daily average ozone concentrations used. For estimating crop damages the measure AOT40c[4] was used for the year 1994. AOT40c data for the individual stations were not available from publications and have been obtained from the air quality laboratory at RIVM directly.

An overview of the number of air monitoring stations for which data were available for the year 1994 is presented in Table 8.3. The average, minimum and maximum concentrations of the individual pollutants across the station measurement data are presented in Table 8.4.

For estimating the impacts the focus is not on the measurements at individual stations but rather on the concentration distribution over the country. To obtain these maps individual station data are spatially interpolated.

RIVM uses a specially developed interpolation method for different pollutants (RIVM, 1996). Comparison of results obtained using different

Table 8.4 Average, maximum and minimum air pollution concentrations measured at the LML stations for 1994

Pollutant	unit	Regional		City		Street	
		average	range	average	range	average	range
SO_2	μg/m^3	7.9	(4.0–20)	14.2	(10.0–22.0)	12.6	(8.0–22.0)
NO_x	ppbV	31.6	(14.2–52.2)	70.9	(60.1–83.9)	135.1	(8.3–273.7)
Particles: PM_{10}	μg/m^3	54.2	(48.1–65.0)	42.8	(36.1–52.4)	36.1	
O_3 (annual average)	μg/m^3	41.8	(36.0–50.0)	32.8	(28.0–40.0)	27.4	(29.9–44.1)
O_3 (six-hour average)	μg/m^3	38.6	(31.0–48.0)	39.0	(37.0–41.0)	41.0	
AOT40c	ppmVh	9.0	(4.1–14)	–	–	–	(23.0–34.0)
CO	mg/m^3	0.39	(0.31–0.44)	0.62	(0.56–0.66)	1.1	
As	ng/m^3	3.8	(1.1–8.8)	–	–	1.0	(39.0–43.0)
Cd	ng/m^3	0.53	(0.34–0.75)	–	–	0.55	
Pb	ng/m^3	28.0	(17.0–36.0)	–	–	44.0	–
Zn	ng/m^3	50.3	(46.0–55.0)	–	–	56.0	(0.63–1.8)

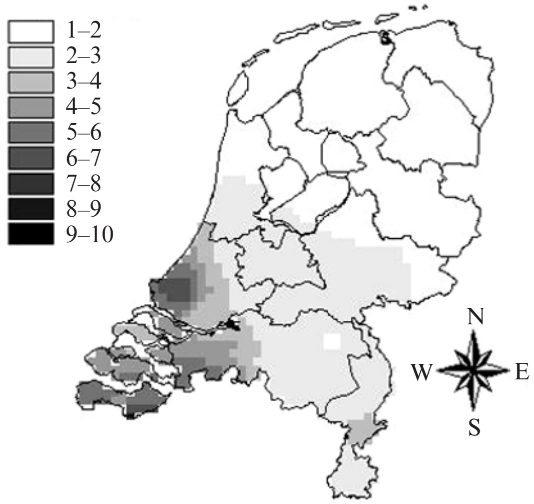
Sources: Airbase (1998), RIVM (1996).

interpolation methods has shown that they differ among each other within a range of 10 per cent (personal communication with members of RIVM's Air Quality Department). A modelling exercise for PM_{10} for 1992 using the TREND model shows a considerable underestimation of actual concentrations. The estimated annual mean concentration was approximately 30 $\mu g/m^3$ (including a 10 per cent correction for wind-blown dust, sea salt aerosol and so on), with a range of 25–35 $\mu g/m^3$, whereas the measured annual mean concentration was 44 $\mu g/m^3$ (average of the regional stations) (RIVM, 1994). RIVM therefore increases all measurement results by 25 per cent (RIVM, 1996). Given these considerations, relying on monitoring data and on a simple interpolation method can be justified for the Dutch situation. Experience within the RIVM, which also uses interpolation methods to construct spatial distribution of pollutant concentrations, has shown that simple nearest neighbourhood interpolation produces credible results (personal communication with members of RIVM's Air Quality Department (Beck, 1998; Diederen, 1998)). The spatial interpolation is therefore performed by linear interpolation of the regional station measurements. Interpolation results are not available for all pollutants (for example, O_3) at RIVM. For these pollutants the simple interpolation technique described above is applied as well.

The interpolation with the available regional stations measurement data was performed using the nearest neighbourhood method in the Geographical Information System ArcView 3.0 (ESRI, 1996). Interpolations were performed on a 1×1 km^2 grid scale. Since most of the stock at risk (population, crops and materials) data were available at the municipality level, the 1×1 km^2 interpolation results were then averaged over the municipality by means of area weighting. Finally, for municipalities with a population density larger than 1500, the average of the city measurement station data was used, except for PM_{10}. For PM_{10} there is no indication from the LML network that city concentrations are significantly above regional concentrations. However, along streets the PM_{10} concentrations can be higher than the background. Street measurement data were not used for the impact analysis, as stock at risk data are not available at this detailed level. Two examples of the resulting maps for 1994 SO_2 concentrations and AOT40c levels are presented in Figures 8.9 and 8.10 respectively.

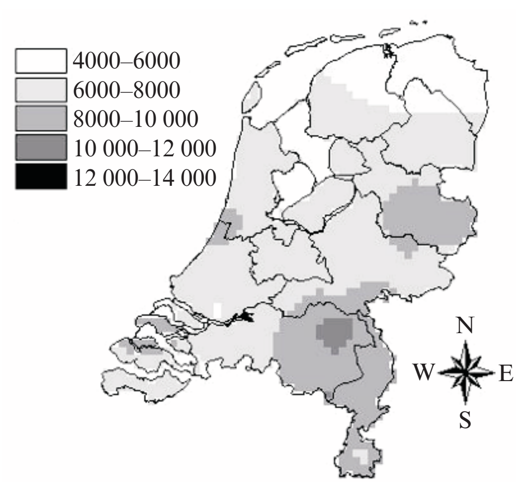
The United Kingdom

Air quality data covering the whole UK have been obtained from three different sources. Data for sulphur dioxide, PM_{10} and ozone were provided by Stedman (1998) at a resolution of 1×1 km. These maps were derived



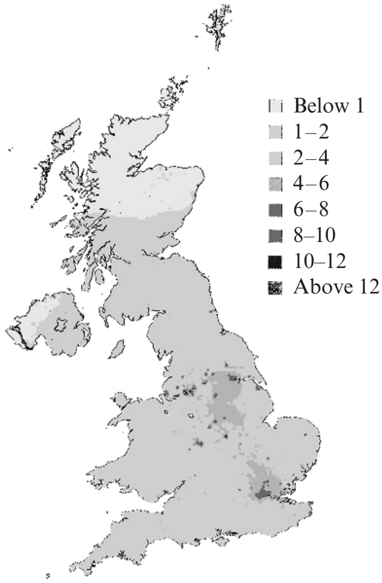
Note: SO_2 concentration in ppb. V.

Figure 8.9 Interpolated 1994 measured SO_2 concentrations based on 26 LML stations in the Netherlands



Note: AOT40c (ppb V.h).

Figure 8.10 Interpolated measured AOT40c levels for 26 regional LML stations in the Netherlands



Note: This map has been plotted at a resolution of 1 × 1 km.

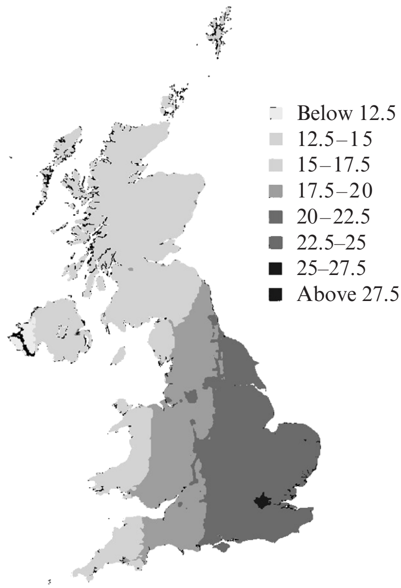
Figure 8.11 Map of annual mean levels SO_2 (ppb) for the UK

from data collected at monitoring sites and surrogate statistics such as emissions from vehicles or population densities. Data for acid deposition were obtained from Vincent et al. (1998). These data were at a resolution of 20 × 20 km. Lastly, ozone AOT40 data were obtained from ITE (Coyle, 1998) at a resolution of 1 × 1 km.

The sulphur dioxide (SO_2) map for 1996 (Figure 8.11) has been derived from SO_2 measurements at automatic monitoring sites and data on domestic emissions. The majority of the monitoring sites are in either rural or city centre locations, and therefore mapped levels are more accurate here than in smaller towns and suburban areas.

Fine Particulate Matter (PM_{10}) is a complex pollutant to map because of the range of sources of both primary and secondary particles that contribute to ambient concentrations. The primary and secondary fractions are therefore estimated separately and added together to produce a map of total PM_{10} for 1996. The secondary particulate fraction of the PM_{10} map was derived from particulate sulphate measurements.

Primary particulate concentrations were calculated at monitoring sites as measured PM_{10} minus estimated secondary particulates. The relationship between primary particulates and local vehicle emissions was then assessed



Note: This map has been plotted at a resolution of 1 × 1 km.

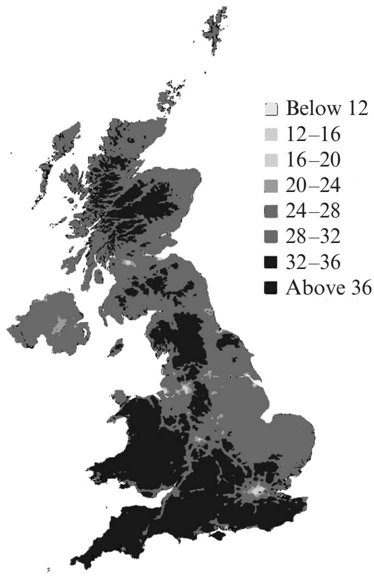
Figure 8.12 Map of the annual mean PM_{10} levels (μg/m³) for the UK

in order to establish the amount of primary particulate not accounted for by these emissions. The other sources of primary particulates are stationary combustion and non-combustion sources. From rural measurement sites it was found that there is an increase in concentration of PM_{10} from these 'other' sources from west to east across the country. A simple linear increase was therefore estimated as the background primary particulate concentration.

The final PM_{10} map for England, Scotland and Wales was produced by adding together the background primary concentration, the vehicle emission related primary concentration and the secondary particulates (see Figure 8.12).

The map for Northern Ireland was produced in a different way, based on SO_2 emission, because PM_{10} concentrations in Northern Ireland are more strongly influenced by domestic heating than vehicles, because of local high use of coal.

The map of *ozone* (O_3) concentrations was generated by estimating separately urban and rural ozone concentrations (Stedman, 1998). Ozone concentrations in urban areas are often lower than concentrations in surrounding rural areas. Estimates of NO_x emissions were used as a surrogate



Note: This map has been plotted at a resolution of 20 × 20 km.

Figure 8.13 Map of the summertime 24-hour mean O_3 levels (ppb) for the UK

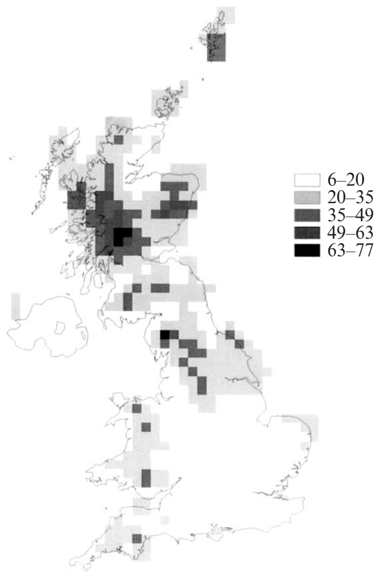
measure of urban ozone levels, and rural levels were estimated using interpolated measures of ozone in rural locations, with altitude as an additional factor in the calculation as levels are higher at higher altitudes. The map generated by Stedman (1998) was a map of the summer 24-hour mean in 1995 (see Figure 8.13). The measure required in the calculation of damages to health is the annual six-hour mean. The map was converted to an annual 24-hour mean using the following equation:

$$\textit{annual mean (ppb)} = (0.95 * \textit{summer mean (ppb)}) - 2$$

To convert from 24-hour mean to six-hour mean the result was multiplied by 1.25.

The crop damage calculations required ozone data as AOT40. This map was obtained from ITE (Coyle, 1998). In general the AOT40 map shows a north to south incline, with relatively higher levels at higher altitudes.

For the UK *acidity* (H^+) was also represented as a map of total wet deposited hydrogen ions – in 1996 this was available at a 20 × 20 km resolution (Figure 8.14). This map has been produced using a 'seeder-feeder'



Note: This map has been plotted at a resolution of 20 × 20 km.

Figure 8.14 Map of annual acid (H^+) deposition for the UK

enhancement, in which upland wet deposition of non-sea-salt sulphate and other ions is up to five times greater than that in lowland areas. The amount of deposition is derived from a combination of sulphur deposition at 32 monitoring sites, rainfall data and an altitude map.

8.2.2 Background Concentrations

The objective of the current study, to relate environmental impacts to human activity, dictates that the damages resulting from emissions of pollution are compared to the level that would exist in the absence of human activity. For some pollutants an estimate of these levels can be gauged from the concentrations measured in remote areas.

The most appropriate data for SO_2 and NO_x levels in northern Europe seems likely to come from north-west Scotland. This region is remote from major industrial activity and the prevailing south-westerly winds are unlikely to carry a high pollution load from other countries. The Review Group on Acid Rain (RGAR, 1990) provides values of about 1 ppbV SO_2 and 2 ppbV NO_2 for this region. Data modelled for the UK using the Harwell Trajectory Model agree well with these figures. Background levels

Table 8.5 Composition of atmospheric particles <2.5 μm in diameter by source, collected in Leeds in 1982/83

Component	Major source	% of total
Ammonium	Natural ammonia emissions	6
Nitrate	Energy sector	5
Sulphate	Energy sector	18
Chloride	Sea salt, coal burning	6
Base cations	Sea salt, dusts	6
Carbonaceous matter	Smoke emissions	37
Insoluble minerals	Wind-blown dust	22

Source: QUARG (1996).

of acid deposition (H^+) for the UK study were estimated from a 20×20 km rainfall map for the UK and the assumption of pH5 rain in pristine areas (RGAR, 1990).

Background PM_{10} concentrations are more difficult to predict, because of diverse sources of particulates of natural origin. These background concentrations are shown in Table 8.5. The sources of fine particulate matter (which is thought to cause most health damages) include natural emissions and agricultural emissions in addition to energy-related emissions, though it is suggested that about 70 per cent of the total arises from human activity. For the assessment here, we used an estimate of PM_{10} natural background of 10 $\mu g\ m^{-3}$.

Ozone is also problematic as background concentrations vary with altitude, sunlight and other climatic conditions. The annual mean 'natural' concentrations assumed here were between 10 and 20 ppbV. Using the conversion factor of 1.25 (see air quality data for the UK in Subsection 8.2.1), these were translated to a range of 16 to 24 ppbV for the annual mean of hourly maximum levels for each day used in the health assessment.

Given the uncertainty concerning the natural background concentrations, in each case a range of background levels is taken into account. Additionally, if not already included in the range as lower bound, the background of zero concentration was applied for the pollutants to estimate the overall damages caused by the respective measured ambient concentration. The background concentrations are listed in Table 8.6. The mid values of the ranges are the recommended values and are used for the summary results. Background values for further pollutants used in the sensitivity analysis for the UK are described in Section 8.4.

Table 8.6 Natural background concentration levels for the studied pollutants

Pollutant	Unit	Low	Mid	High
SO_2	ppbV[a]	0	1	2
Particles: PM_{10}	$\mu g/m^3$	5	10	15
O_3[b]	ppbV[c]	16	20	24
AOT40c	ppbV.h	0	0	0
CO	ppmV[d]	–	0.15	–
Heavy metals: As, Cd, Pb, Zn	ng/m^3	0	0	0
Heavy metals: Cr, Hg	–	0	0	0

Notes:
[a] 1 ppbV $SO_2 = 2.63\ \mu g/m^3$.
[b] annual average of daily six-hour average (from sunshine peak hours).
[c] 1 ppbV $O_3 = 2\ \mu g/m^3$.
[d] 1 ppbV $CO = 1.163\ \mu g/m^3$.

8.2.3 Stock at Risk Data

Different databases and methods were applied to get information about the geographical distributions of population, crops and materials in Germany, Italy, the Netherlands and the UK. In the following the data sources for the four countries are individually described. In general, 1994 was used as a reference year.

Germany

The receptor data applied for the German study were all taken from the database of the EcoSense programme system. These were implemented for European damage estimations and have already been applied for many calculations, for example, in the ExternE project (European Commission, 1995, 1997b). A more detailed description of the applied data follows (see also Mayerhofer et al., 1997).

Population data and data on *crop cultivations* for 1993/94 are available on NUTS levels from the REGIO database published by Eurostat (Eurostat, 1995). These were implemented and used for the damage estimations. In addition, the percentages of different risk groups are needed for the estimation of damages. Specific risk group fractions are applied for the German part of the study (Statistisches Bundesamt, 1995). These are: male (49 per cent); female (51 per cent); adults – people older than

18 years – (80 per cent); people older than 30 years (56 per cent); people older than 65 years (13 per cent); and asthmatics (3.5 per cent).

Material data were derived based on building identikits of the cities Dortmund and Cologne (Hoos et al., 1987). For the eastern part of Germany these were extrapolated by using population data, while for the western part the results of the building census of 1987 could be applied (Statistisches Bundesamt, 1987), as indicated by the ExternE methodology, establishing relationships between population and materials (European Commission, 1998).

Italy

For Italy only population and crops were considered as stocks at risk. *Population* data for the year 1994 were derived from a specific publication of the Italian Statistical Office on population published in 1996 (ISTAT, 1996). Population distribution was then calculated on the basis of the EcoSense programme system.

For crop data we referred to the same statistics used for the GARP I project, which were reported in a specific publication of the Italian Statistical Office on agriculture (ISTAT, 1991).

The Netherlands

Data on the *population* on 1 January 1993, by age (five-year age groups) and sex per municipality were obtained from the Central Bureau of Statistics (CBS, 1998). For the Netherlands, 1993 data were used because the GIS municipality map, needed for estimating the pollutant concentrations in the municipalities, was available for 1993 municipality boundaries only (as the municipality boundaries, names and number changed in 1994). The population growth between 1993 and 1994 is of the order of 2 per cent while no significant migration or shifts in age distribution over the different municipalities occurred. The error introduced by using 1993 data for the 1994 analysis is therefore small compared to the uncertainty in the exposure–response functions and the valuation of the health impacts. The total population on January 1993 corresponded to 15 237 463 inhabitants and was classified as follows: 82 per cent of adults (15+ years old), 3 per cent of children (under two-years old), 13 per cent of elderly people (65+ years old), 51 per cent of female and 49 per cent of male people.

Mortality data on municipality level can only be obtained through analyses of postal code-based data. These data, collected by the Central Bureau of Statistics, are not readily available on grounds of private confidentiality.

The mortality data on province and regional public health service (GGD) level were, however, available from a Regional Public Health Profile database made by the National Institute for Public Health and Environmental Protection (RIVM, 1990). These data are average figures for mortality due to a number of causes for the period 1985–89. More recent data are not available at this detailed level. Mortality due to accidents and violent causes can be subtracted from the total mortality figures. The resulting mortality rates for non-violent death at the provincial level range from 0.760 per cent to 0.949 per cent for men, and from 0.726 per cent to 0.809 per cent for women. These mortality rates were used for the calculations. Each municipality was assigned the mortality rate of the province it belonged to.

Some of the exposure–response functions for morbidity impacts have to be applied to a certain subset of the general population, such as children or asthmatics. The percentage of asthmatics is estimated at 1.7 per cent of the population over 18 years of age according to figures for 1986 (CBS, 1989). Later figures are not available because since 1987 only an aggregate estimate for asthma, chronic bronchitis and CARA (chronic a-specific respiratory diseases) has been provided by CBS. We have applied the adult-percentage to all age groups. This may result in an underestimate of the number of asthmatics since asthma is more prevalent in children than in adults. Exposure–response functions for children were applied to people under 15 years, whereas exposure–response functions for adults were applied to all persons from 15 years of age.

Data on area cultivated of each *crop* in 1993 was obtained from the Dutch Central Bureau of Statistics (CBS, 1998). Though the base year is 1994, 1993 data are used in this analysis for the same reason as stated above in the section regarding population. The crops cultivated in the different agricultural areas can differ significantly between 1993 and 1994. For the most common crops (wheat, barley, rye and oats) the average amount of crops grown in the Netherlands are within a range of 25 per cent difference between 1993 and 1994. The error introduced by using 1993 data for the 1994 analysis is therefore relatively small compared to the uncertainty introduced when using crops' exposure–response functions to crop damages. For each municipality in the Netherlands data on the area cultivated (ha) were available. Average crop yield per ha is estimated each year by special committees, and is published by the CBS (CBS, 1998). By combining area cultivated per municipality (in ha) with the estimated average yield per ha, the yield per municipality can be calculated. Table 8.7 shows the estimates on total yield obtained for the crops studied.

Data on the *material stock* for each municipality was not available. Therefore, for the Netherlands also these were estimated using primarily

Table 8.7 Estimated total yields in 1993 in the Netherlands

Crop	Area cultivated (1000 ha)	Estimated yield (ktonne)
Winter wheat	99.8	888
Summer wheat	18.2	142
Winter barley	4.4	29
Summer barley	35.7	225
Rye	7.4	42
Peas and beans:		
Fresh harvested green pea[1]	6.6	29
Dry harvested green pea	2.2	9.5
Marrow pea	0.95	4.5
Brown beans	2.4	6.6
Field beans	1.3	7.0
Oats	5.2	31
Potato:		
Seed-potato grown on sand/peat	5.5	235
Seed-potato grown on clay	53.0	1050
Consumption potato grown on	14.4	663
sand/peat	62.0	2770
Consumption potato grown on clay	62.9	2950

Note:
[1] No specific data on yield per ha available; the yield per ha of dry harvested peas is used.

Source: CBS (1998).

the relations between stock of materials and population, as described in the ExternE methodology and defined in the EcoSense 2.0 model (IER, 1997). These relations are available for zinc, galvanised steel, sandstone, limestone, natural stone, mortar rendering and paint. Gosseling et al. (1990) made similar stock estimates for the Netherlands for zinc, galvanised steel and painted metals. For zinc the two estimates gave similar results, whereas for galvanised steel the difference was much larger. The difference is probably due to the rough stock estimation applied in ExternE methodology while the Gosseling study took a more detailed approach. For 'paint' the Gosseling study only identified painted metals while the other study also includes other painted objects. Therefore, all relations (for zinc, sandstone, limestone, natural stone, mortar, rendering and paint) were taken from the ExternE methodology with the exception of galvanised steel for which the relation from the study by Gosseling et al. was applied.

United Kingdom

Population data was taken from the 1991 Census data for the UK, with post-code level data re-gridded into 1 km^2 cells.

For *crops*, land use data were categorised into areas within each 150 km grid square of permanent crops, arable crops, grassland and forest. Crop production data for the UK (Philips, 1995) were converted to production values at world prices using data from the Food and Agriculture Organisation (FAO, 1994) then allocated across these grid cells in proportion to the areas of different land use types occurring. This gave a spatially disaggregated stock at risk in terms of total production value.

The *materials stock* at risk is derived from data on building numbers and construction materials taken from building survey information. The UK stock at risk was derived from Butlin et al. (1994), which provides a summary of typical building identikits, for a range of modern buildings, by region of the UK, along with estimates of the numbers of each building. This regional disaggregation reflects the variations in the type of construction and materials used by area. The materials for which damage has been considered are calcareous stone, mortar, paint, concrete, aluminium and galvanised steel. Although not exhaustive, this list includes the most sensitive of the materials commonly used by the construction industry. All steel is assumed to be painted and this stock is therefore transferred into the paint inventory. A separate inventory was compiled for galvanised steel, which included other categories as well as buildings (European Commission, 1995). Within each region, the distribution of materials was disaggregated by population. The UK materials inventory for each 1 km^2 grid cell was then calculated by disaggregating building material relative to population distribution. Reference meteorological data (precipitation, humidity, and so on) at a resolution of 100 km^2 were used.

8.3 RESULTS OF IMPACTS ON HEALTH, CROPS AND MATERIALS

This section deals with the presentation of the results for each of the four countries that are obtained using the EcoSense model on the basis of the air quality data described in subsection 8.2.1. Monetary damages are presented in this section, whereas detailed physical impacts are to be found in the Appendix to this chapter.

8.3.1 Germany

Human health damages – morbidity

The effects on morbidity caused by SO_2, fine particles (PM_{10}) and ozone are presented in the Appendix (Tables A.8.1, A.8.2 and A.8.3 respectively). The largest impact is caused by PM_{10}, which is also the only pollutant responsible for real chronic effects: new cases of chronic bronchitis in adults. Therefore the effects of PM_{10} are expected to dominate the monetary damages of morbidity. The comparison of the respiratory hospital admissions, which is the sole impact estimated for all three pollutants, shows that O_3 causes the largest damages corresponding to 4700 to 9300 cases.

The estimated number of cases of cancer caused by human health exposure to cadmium due to inhalation are very small, as Table A.8.4 in the Appendix shows. The results were estimated by applying the recommended unit risk factors for the concentration of 27.5 pg/m^3 cadmium which corresponds to the rough estimates of the annual mean concentration in Germany for 1994. Even if it is assumed that each case of cancer leads immediately to death (impacts valued by the value of statistical life of 3.1 million Euro), which gives an impression of the maximum possible damage costs, the estimated damage costs are only about 1.8 million Euro per annum, which is a very small amount compared with damage costs for chronic bronchitis, for example, caused by PM_{10} at 16 billion Euro per annum (estimated using the recommended monetary value of 240 000 Euro per case). Because of their limited amount and because of their high degree of uncertainty, these costs are neglected in the following.

Human health damages – mortality

The effects on mortality can be subdivided into chronic and acute health effects. While for SO_2 and O_3 only acute effects could be estimated because of the lack of chronic effect data, 99 per cent of particles mortality impacts are constituted by chronic effects. Tables A.8.5 to A.8.7 in the Appendix give evidence that many more life-years are lost because of chronic rather than acute effects. This is also why the damage costs caused by PM_{10} are expected to be larger than the costs caused by O_3 and SO_2.

Monetary damage of both chronic and acute mortality effects are illustrated in Table 8.8. Two different discount rates (3 per cent and 11 per cent) are used here for the analysis of mortality damage costs. The damages on mortality caused by PM_{10} for a discount rate of 3 per cent – consisting of about 96 per cent chronic effects – are much larger than the damage costs due to acute effects of SO_2 and O_3. At a discount rate of 11 per cent the damage costs of exposure to SO_2 and O_3 are of the same order of magnitude as the costs caused by PM_{10} (75 per cent chronic effects).

Table 8.8 Estimated damage costs for acute and chronic effects on mortality for the discount rates of 3 and 11 per cent in Germany in 1994

Pollutant	Mortality (billion Euro p.a.)	
	Discount rate	
	3%	11%
SO_2 (background 1 ppbV) Acute YOLL	0.74	1.6
O_3 (background 20 ppbV) Acute YOLL	1.3	2.7
PM_{10} (background 10 μg/m^3) YOLL	20 (96% chronic YOLL)	8.3 (75% chronic YOLL)
Total damage costs	22	13

Crop damages

For crops both positive and negative effects can be observed. In fact, a yield increase compared to natural baseline pollution conditions was observed at low concentrations of measured SO_2 that are associated with a dominant fertilising effect. However, the positive effects caused by SO_2, even for an assumption of zero background, are much smaller than the yield losses of crops due to ozone concentrations. The largest yield loss was observed for potatoes at about 5800 kt/a in 1994, as shown by Table A.8.8 in the Appendix.

Material damages

Effects on paints clearly dominate material damages. The largest surface areas which had to be repaired in 1994 because of anthropogenic changes in SO_2 concentration were about 6 to 10 km^2 of painted surfaces, followed by 0.39 to 0.66 km^2 of rendering surfaces. Large impacts are also observed for galvanised steel (0.19 to 1.3 km^2) and zinc (0.026 to 0.18 km^2) which are affected by O_3 and SO_2. Data are summarised in the Appendix in Table A.8.9.

Damage costs estimated for crops, material and human health

Tables 8.9 to 8.12 show the damage costs caused by the different pollutants estimated from the described impacts by using the recommended monetary values (see Chapter 5). The total damage costs due to fine particles are

Table 8.9 Estimated damages caused by PM_{10} in Germany in 1994

Function	Damages (million Euro p.a.)			
	Background			
	0 μg/m³	5 μg/m³	10 μg/m³	15 μg/m³
Mortality (3% discount rate)	30 000	25 000	20 000	15 000
Morbidity	29 000	24 000	20 000	15 000
Total damage costs	59 000	49 000	39 000	30 000

Table 8.10 Estimated damage costs caused by synergetic effects of O_3 and SO_2 in Germany in 1994

Function	Damages (million Euro p.a.)			
	Background: SO_2/O_3 (6-hr mean)			
	0/0 ppbV	0/16 ppbV	1/20 ppbV	2/24 ppbV
Material	37	19	11	5.3
Total damage costs	37	19	11	5.3

Table 8.11 Estimated damage costs caused by SO_2 in Germany in 1994

Function	Damages (million Euro p.a.)		
	Background		
	0 ppbV	1 ppbV	2 ppbV
Material	160	120	92
Crops	−99	−59	−26
Morbidity	18	15	12
Mortality (3% discount rate)	910	740	570
Total damage costs	990	820	650

estimated at about 39 billion Euro per annum with a range of 30 to 49 billion Euro per annum for estimated natural background concentrations from 5 to 15 μg/m³. Damage costs due to mortality effects caused by PM_{10} estimated for a discount rate of 3 per cent are slightly higher than the

Table 8.12 Estimated damage costs caused by O_3 in Germany in 1994

Function	Damages (million Euro p.a.)			
	Background			
	0 ppbV	16 ppbV (6-hr mean)	20 ppbV (6-hr mean)	24 ppbV (6-hr mean)
Crops	1700	1700	1700	1700
Morbidity	3400	1700	1300	860
Mortality (3% discount rate)	4800	2400	1800	1200
Total damage costs	9900	5800	4800	3800

respective morbidity effects. These costs are significantly higher than the total costs derived for O_3 and SO_2 concentrations that correspond to 4.8 (range: 3.8 to 5.8) and 0.82 (range: 0.65 to 1.0) billion Euro per annum respectively. The damage costs estimated for combined effects of SO_2 and O_3 on zinc and galvanised steel at 0.01 billion Euro per annum are negligible in this context. The damage costs of ozone on crops (1.7 bn Euro p.a.), morbidity (1.3 bn Euro p.a.), and mortality (1.8 bn Euro p.a.) are comparable, whilst the damage costs caused by SO_2 differ greatly between the various categories: 120 million Euro per annum for materials, 59 million Euro per annum for crops, 15 million Euro per annum for morbidity and 740 million Euro per annum for mortality.

It can be seen that by far the largest damages are associated with PM_{10}. The largest effects, representing 98 per cent of the PM_{10} damage costs, are represented by costs for new cases of chronic bronchitis in adults (chronic morbidity) at 16 billion Euro per annum, chronic mortality at 20 billion Euro per annum, and restricted activity days at 2.6 billion Euro per annum.

Conclusions

The German study includes damage costs estimated for impacts of SO_2, PM_{10} and O_3 on crop yields, building material and human health as well as a rough estimation of health damages caused by cadmium in air in 1994.

The damage estimation includes uncertainties concerning interpolation, background levels and, especially for human health impacts, the exposure–response functions and monetary valuation. Not all effects caused by the pollutants could be taken into account. For instance, for material damages, the soiling effects of PM_{10} and the impacts of O_3 on rubber were not assessed. Also damages on ecosystems are not included because processes in these ecosystems are so complex that a description of the impacts of

Table 8.13 Estimated damage costs caused by all pollutants in Germany in 1994

Function	Damages (million Euro p.a.) Pollutant (Natural background)			
	O_3 (20 ppbV)	SO_2 (1 ppbV)	PM_{10} (10 $\mu g/m^3$)	Total Damage Costs
Material	N.A.	120 + 10[a]	N.A.	130
Crops	1700	–59	N.A.	1600
Morbidity	1300	15	20 000	21 000
Mortality (3% discount rate)	1800	740	20 000	22 000
Total damage costs	4800	830	39 000	45 000

Notes:
N.A.: not assessed.
[a] damage costs caused by O_3 and SO_2.

exposure to pollutants in exposure–response relations as well as monetary evaluation is not yet possible. However, these effects may be important, because the critical levels reported for natural vegetation and forest ecosystems are exceeded in many regions of Germany in 1994. Also, given that these impacts have been the driving force behind a lot of pan-European legislation on reduction of trans-boundary air pollution, it would seem that a high valuation is given to them implicitly by decision makers as well as members of the public. Therefore, much effort is needed in the investigation of exposure–response relations and monetary valuation in the future.

Table 8.13 shows the damage costs estimated for the pollutants O_3, SO_2 and PM_{10} in this study by assuming the recommended natural background concentrations. The contribution of PM_{10} to damage costs estimates for health effects clearly dominates the results. This corresponds to 93 per cent of total health effects and is mainly (98 per cent) caused by three impacts, namely effects of chronic exposure on mortality and chronic bronchitis, and effects of acute exposure on incidence of restricted activity days.

While 87 per cent of the total damage costs are caused by PM_{10} (corresponding to 45 billion Euro p.a.), only 11 per cent are caused by O_3 and 2 per cent by SO_2. The rough estimate for maximal damage costs due to cadmium exposure is not included in the table, because the estimated costs of 1.8 million Euro per annum are, comparatively, negligible. The damage costs on human health at 43 billion Euro per annum correspond to 96 per cent of the total damage costs. Only about 4 per cent of the total costs are damage costs estimated from impacts on building material, while the effects on yield crops represent about 0.3 per cent.

8.3.2 Italy

Human health damages

For human health, damages caused by SO_2, PM_{10} and O_3 concentrations over different receptors were calculated. Background values of 1 ppbV, 10 $\mu g/m^3$ and 20 ppbV were considered for the final impacts calculations of SO_2, PM_{10} and O_3 respectively. Impacts and damages have been calculated for the concentration exceedance only. The results are given in Tables 8.14 to 8.16.

Due to the low spatial coverage of O_3 concentrations, an average concentration value has been used for the whole country, rather than provincial values obtained with interpolation (see for explanations the part of Section 8.1.2 relating to Italy). Some indications about the error introduced using this approximation are obtained when considering that damages calculated using an average concentration for the whole country are 10 to 20 per cent lower than the ones obtained with provincial concentration values for PM_{10} and SO_2 (the difference depends on the chosen background value). The reason is that concentrations are generally higher where the population is also higher; differences for ozone should be less important, since O_3 concentrations are not as closely correlated with population as PM_{10} and SO_2. Since the average O_3 concentration calculated from the available measurements (nearly 17 ppbV) is below the background value

Table 8.14 Estimated damages resulting from SO_2 on human health in Italy in 1994

Function		Damage costs (million Euro p.a.)		
		SO_2		
		Background (ppbV)		
Receptor	Impact category	0	1	2
ENTIRE POPULATION				
Respiratory hospital admissions		13	11	9
Acute mortality	3%	700	580	470
	11%	1300	1100	880

Table 8.15 Estimated damages resulting from PM_{10} on human health in Italy in 1994

Function		Damage costs (million Euro p.a.)				
		PM_{10} – Background (μg/m^3)				
Receptor	Impact category	0	5	10	15	20
ASTHMATICS (3.5% of population)						
Adults	Bronchodilator usage	420	370	330	280	230
	Cough	82	73	63	54	45
	Lower respiratory symptoms (wheeze)	32	28	25	21	17
Children	Bronchodilator usage	50	45	39	33	27
	Cough	16	14	13	11	9
	Lower respiratory symptoms (wheeze)	13	12	10	9	7
ELDERLY 65+ (14% of population)						
	Congestive heart failure	51	45	39	33	28
CHILDREN (20% of population)						
	Chronic bronchitis	180	160	140	120	100
	Chronic cough	230	200	180	150	130
ADULTS (80% of population)						
	Restricted activity days	3700	3300	2900	2500	2000
	Chronic bronchitis	23 400	20 800	18 100	15 400	12 800

Table 8.15 (continued)

Function		Damage costs (million Euro p.a.)				
		PM$_{10}$ – Background ($\mu g/m^3$)				
Receptor	Impact category	0	5	10	15	20
ENTIRE POPULATION						
Respiratory hospital admissions		41	36	31	27	22
Cerebrovascular hospital admissions		99	87	76	65	54
Acute mortality	3%	1000	920	800	670	550
	11%	2200	2000	1700	1500	1200
Chronic mortality	3%	28 400	25 200	22 000	18 800	15 500
	11%	11 800	10 500	9200	78000	6500

Table 8.16 Estimated damages resulting from effects of O_3 on human health in Italy in 1994

Function		Damage costs (million Euro p.a.)			
		O_3 – Background (ppbV)			
Receptor	Impact category	0	16	20*	24*
ASTHMATICS (3.5% of population)					
All Asthma attacks		100	6	0	0
ADULTS (80% of population)					
Minor restricted activity days		110	6	0	0
ENTIRE POPULATION					
Respiratory hospital admissions		52	3	0	0
Symptom days		460	26	0	0
Acute mortality	3%	1200	65	0	0
	11%	2500	137	0	0

Note: * no damage is reported, because the average O_3 concentration is below the background value.

(20 ppbV) considered for the purpose of our analysis, no damage from O_3 has been reported in the remainder of the analysis.

Sensitivity analyses have been performed in order to estimate the influence of the choice of the background concentrations on the results. Values of 0 and 2 ppbV for SO_2; 0, 5, 15 and 20 $\mu g/m^3$ for PM_{10}; 0, 16 and 24 ppbV for O_3 concentrations have been considered. Results of the sensitivity analyses are reported separately for SO_2, PM_{10} and O_3. Impacts are calculated both in terms of cases per year for each endpoint as presented in the Appendix (Tables A.8.10 to A.8.12) and in monetary terms as summarised in Tables 8.14 to 8.16. As for damage costs results, both 3 and 11 per cent are used as discount rates for mortality estimates.

Crop damages

Damages to barley, potato, sugar beet, wheat, oats and rye due to SO_2 exposure have been calculated using the exposure–response functions of Baker et al. (1986), modified in order to account for the fertilising effect of low SO_2 concentrations. As for O_3 damage, no calculation has been made, because the available data did not allow the calculation of AOT40 values.

Results were calculated against a 1 ppbV background value for SO_2 concentrations; sensitivity analyses have been performed with background

Table 8.17 Estimated damage resulting from effects of SO_2 on crops in Italy in 1994

Function	Damage costs (kEuro p.a.)			
	SO_2			
	Monetary value		Background	
	(Euro/t)	0 ppbV	1 ppbV	2ppbV
barley	5.4	220	160	100
oats	5.6	2	2	1
potato	8.2	350	230	130
rye	15.6	130	88	58
sugar beet	4.8	1,100	770	460
wheat	9.6	1,600	990	500

Source: Baker et al. (1986), modified.

values of 0 and 2 ppbV respectively, in order to investigate the differences in results obtained using national rather than regional production values. Damages calculated for potato, sugar beet and wheat with the national production value are respectively 28, 17 and 44 per cent higher than the ones calculated with the regional productions. This means that SO_2 concentrations, as expected, are generally lower in rural regions, where crops are present. Moreover, this difference drops or rises as soon as the SO_2 background concentration is reduced or raised. This should not be a great problem, since the Italian production of barley, oats and rye is quite low, but it should be taken into account that results for these crops could be slightly overestimated. Table 8.17 summarises the results of the calculations in monetary terms (and Table A.8.13 in the Appendix reports the results in physical terms).

Conclusions

Table 8.18 summarises the cost of estimated atmospheric pollution damages to crops and human health in Italy in 1994. As for materials, no calculation has been made, since no data were available on receptors distribution, or wet acid deposition, which are required for these calculations.

Most of the damages (about 98 per cent) arise from PM_{10} effects on human health, almost equally distributed between mortality and morbidity. It is noteworthy that the exposure-response function from Abbey et al. (1995) for chronic bronchitis in adults accounts for more than 80 per cent of the total morbidity damage, and the function from Ostro (1987) for restricted activity days accounts for another 13 per cent. Damage to crops

Table 8.18 Summary of atmospheric pollution damages in Italy in 1994

Impact category	Damage costs (million Euro)			
	SO_2	PM_{10}	O_3	Total
Material	N.A.	N.A.	N.A.	N.A.
Crops	2	N.A.	N.A.	2
Morbidity	11	22 000	0	22 000
Mortality (3% discount rate)	580	22 000	0	23 000
Total damage costs	590	44 000	0	45 000

Notes:
Figures for SO_2 based on 1 ppbV background level.
PM_{10} figures based on 10 g/m^3 background level.
O_3 figures based on 20 ppbV background level.

is only a small percentage of the total, and is negligible in comparison to the total damage.

Further research is needed on ozone concentrations in Italy, in order to calculate AOT40 for crop damages and, more importantly, to evaluate ozone concentrations at an appropriate level of spatial disaggregation. In fact, with zero background concentration values for SO_2, PM_{10} and O_3, ozone related morbidity damage is more than 50 times higher than for SO_2, and mortality damage is almost double the value for SO_2, and it is higher than the acute mortality value related to PM_{10}. Therefore, a correct evaluation of ozone concentrations is considered important.

8.3.3 The Netherlands

This section presents the results of the impact and damage estimates due to air pollution concentrations exceeding background concentrations in the Netherlands in 1994.

Human health damages

Tables A.8.14 to A.8.16 in the Appendix provide results for morbidity and mortality impacts in terms of number of cases caused by concentrations of air pollutants (PM_{10}, SO_2, O_3, CO, As and Cd) in excess of given natural background levels. Chronic mortality impacts, due to PM_{10} concentrations above different background levels, are only presented in physical terms assuming that there is an average delay of 10 years in the impact (see Part I).

The valuation of the morbidity and acute mortality impacts is performed by using single monetary values per unit of impact type. The chronic mortality impacts are valued by using age-dependent willingness-to-pay data. The estimated damages for all morbidity and mortality impacts are presented in Tables 8.19 to 8.22 and are summarised in Table 8.23.

The total number of deaths in the Netherlands in 1994 was 133 471. Of these deaths some 5500 were attributed to chronic obstructive pulmonary disease (COPD), 12 000 to respiratory diseases and 38 000 to ischaemic heart and cerebrovascular diseases (CBS, 1998). Acute mortality in consequence of air pollution above the recommended natural background corresponds to about 2700 additional deaths. This corresponds to 2.0 per cent of all deaths and to 4.9 per cent of all respiratory and cardiovascular deaths, which seems a plausible estimate.

Regarding chronic bronchitis this study estimates 17 000 additional cases in the Netherlands due to PM_{10} air pollution in 1994 above the recommended natural background of 10 $\mu g/m^3$. As mentioned before, 1.7 per cent of the Dutch population aged 18 years and older suffers from chronic bronchitis (CBS, 1989) which corresponds to about 110 000 adults. Therefore, the additional cases estimated here are some 15 per cent of the total number of adults suffering from chronic bronchitis. This seems to be an overestimation of the real impact. Due to the high monetary valuation of new chronic bronchitis cases (240 000 Euro per case) the associated monetary damage cost due to PM_{10} concentrations above the recommended natural background level (10 $\mu g/m^3$) is estimated at 4.1 billion Euro in 1994.

A summary of the human health damage costs due to analysed pollutants above the recommended natural background levels (10 $\mu g/m^3$ for PM_{10}, 1 ppbV for SO_2, 20 ppbV for O_3, 0.15 ppbV for CO and 0 ng/m^3 for heavy metals) is presented in Table 8.23. Only a subtotal of the damages is given as not all air pollutants are analysed. The subtotal human health damages amount to some 10 billion Euro at the recommended discount rate of 3 per cent. This is 3.9 per cent of Dutch GDP.

Crop damages

Table A.8.17 and Table A.8.18 in the Appendix present the estimated crop impacts in physical terms due to air pollution above different background levels. As mentioned previously in the description of air quality data (Section 8.2.1) the impact from ozone on crops is due to a threshold value called AOT40c being exceeded. The SO_2 exposure–response function for most of the analysed crops is non-linear and the slope of the function is even negative (giving increased crop yield through fertilisation) at low SO_2 concentrations, as explained in the methodology section. As SO_2 concentrations in the countryside were very low in 1994, typically between 1.5 and 7.6 ppbV, the exposure–response function yields benefits for SO_2. It is unlikely that the increased SO_2 levels will have a beneficial fertilisation effect, given the already abundant use of fertilisers.

Rice and tobacco were produced in negligible quantities in the Netherlands, whereas data on the production of sunflower seeds were not readily available for this study. Therefore, these impacts were not analysed.

In this section different monetary values are used for quantifying the damages. They are presented in Table 8.24. The first column lists the damage values recommended in the methodology section. The second and third columns present domestic producer prices and international prices both given by the FAO in 1994 and 1991 respectively. The average landing prices of the crops at Rotterdam harbour (Agricultural Economics Research Institute, The Hague, ExMis Database) are very similar to domestic producer prices and are therefore not listed separately.

The estimated damages using these different prices are presented in Table 8.25. As the impacts are inflicted on Dutch farmers and because landing prices are similar to domestic producer prices, the damage estimates using domestic producer prices are probably most in line with the real situation assuming the impacts are estimated correctly. Thus, the subtotal crop damages due to SO_2 and O_3 air pollution levels above recommended natural background levels (1 ppbV for SO_2 and 0 ppmh for AOT40c) are estimated to be some 154 million Euro, that is, 0.06 per cent of Dutch GDP in 1994, if domestic prices are used. However, the methodology does not consider the fact that farmers will shift to other crops when production is not profitable. Furthermore, prices are not given in real markets but rather depend on supply and demand. A more detailed analysis that takes these factors into account is presented as a case study in Chapter 10.

Material damages

The materials exposure–response functions also include pollutant parameters not analysed in this study (for example, acid deposition, time of wetness and relative humidity parameters). The wet acid deposition (WAD) parameter will change by reducing air pollution concentrations to different natural background levels. The influence of the wet acid deposition parameter in the functions on the extent of the impact is only of the order of a few per cent and is therefore neglected. It is assumed that the WAD is constant for all regions at the average 1994 level (100 meq/m^2.y) (IER, 1997). Furthermore, it is assumed that there is no change in the time of wetness (TOW) and the fraction's relative humidity (FRH) with changes in pollution concentrations to natural background levels. The assumption that these are constant for a small country like the Netherlands is extremely unlikely to lead to significant error. The TOW is set at 0.58 while the FRH is set at 0.41 (IER, 1997).

Table 8.19 Morbidity damages due to SO_2, PM_{10}, O_3 and CO pollution in the Netherlands in 1994

Receptor	Health impact	Damage costs (million Euro p.a.)			
		Background			
		zero	low	mid	high
		SO_2			
		0 ppbV	0 ppbV	1 ppbV	2 ppbV
Entire population	Respiratory hospital admissions	2	2	2	1
		PM_{10}			
		0 μg/m^3	5 μg/m^3	10 μg/m^3	15 μg/m^3
Asthmatics					
Adults	Bronchodilator usage	48	42	36	29
	Cough	9	8	7	6
	Lower respiratory symptoms	4	3	3	2
Children	Bronchodilator usage	5	5	4	3
	Cough	2	2	1	1
	Lower respiratory symptoms	1	1	1	1
Elderly 65+	Congestive heart failure	11	10	8	7
Children 15–	Bronchitis	38	33	28	23
	Cough	49	43	36	30
Adults 15+	Chronic bronchitis	5500	4800	4100	3400
	Net restricted activity days	870	760	640	520

Entire population	Respiratory hospital admissions	9	8	7	6
	Cerebrovascular hospital admissions	23	20	17	14
		O_3			
		0 ppbV	16 ppbV	20 ppbV	24 ppbV
Asthmatics	All asthma attacks	20	7	4	1
Entire population	Respiratory hospital admissions	21	7	4	1
	Symptom days	170	66	35	5
Adults 15+	Minor restricted activity days	44	15	8	0
		CO			
		0 ppbV		0.15 ppbV	
Elderly 65+	Congestive heart failure	4		3	
Subtotal	SO_2	2	2	2	1
	PM_{10}	6600	5700	4900	4000
	O_3	270	95	51	7
	CO	4		3	
Totals	All	6900	5800	5000	4000

Table 8.20 Cancer caused by air pollution due to selected heavy metals in the Netherlands in 1994

Receptor	Health impact	Pollutant	Damage costs (million Euro p.a.)	
			Background level 0 ng/m3	
			Discount rate 3%	Discount rate 11%
Entire population	Cancer	As	7.4E−02	5.6E−03
	Cancer	Cd	1.1E−02	8.0E−04
Subtotal		All pollutants	8.4E−02	6.4E−03

Table 8.21 Acute mortality damages due to PM_{10}, SO_2 and CO air pollution in the Netherlands in 1994

Receptor	Discount rate	Damage costs (million Euro p.a.)			
		Background			
		zero	low	mid	high
		PM_{10}			
		0 μg/m³	5 μg/m³	10 μg/m³	15 μg/m³
Entire population	3%	280	240	200	170
	11%	580	500	430	350
		SO_2			
		0 ppbV	0 ppbV	1 ppbV	2 ppbV
Entire population	3%	120	120	90	55
	11%	260	260	190	120
		O_3			
		0 ppbV	16 ppbV	20 ppbV	24 ppbV
Entire population	3%	530	190	100	15
	11%	1100	400	210	32
Subtotal	3%	930	550	400	240
	11%	2000	1200	830	500

Table 8.22 Chronic mortality damages due to PM_{10} air pollution in the Netherlands in 1994

Discount rate	Delay to one year change	Monetary value[a]	Damage costs (million Euro p.a.)			
			Background			
			zero	low	mid	high
			0 $\mu g/m^3$	5 $\mu g/m^3$	10 $\mu g/m^3$	15 $\mu g/m^3$
3%	0 years	28.2	7800	6700	5700	4700
	20 years	18.1	5000	4300	3700	3000
	average of 10 years	23.2	6400	5500	4700	3900
11%	0 years	16.6	4600	4000	3400	2800
	20 years	2.6	720	620	530	430
	average of 10 years	9.6	2600	2300	2000	1600

Note: [a] Million Euro per 100 000 population for a one-year change of 21 $\mu g/m^3$ PM_{10}.

Table 8.23 Summary of air pollution damages to human health due to air polluants in the Netherlands in 1994 exceeding the recommended mid natural background concentrations (PM_{10} = 10 μg/m3 , O_{3} = 20 ppbV, SO_2 = 1 ppbV and CO = 0.15 ppbV)

Impact	Pollutant	Damage costs (million Euro p.a.)		Damage costs (% 1994 GDP)	
Morbidity	SO_2	2		6.5E-04	
	PM_{10}	4900		1.9	
	O_3	51		0.020	
	CO	3		1.0E-03	
	Subtotal	5000		1.9	
Mortality		Discount rate (%)		Discount rate (%)	
		3	11	3	11
Acute mortality	SO_2	90	190	3.5E-02	7.3E-02
	PM_{10}	200	430	0.079	0.17
	O_3	100	210	0.039	0.083
	As + Cd (cancer)	0.084	0.006	3.3E-05	2.5E-06
	Subtotal	390	830	0.2	0.3
Chronic mortality	PM_{10} (average 10-year delay)	4700	2000	1.8	0.8
Subtotal		10 000	7800	3.9	3.0

The estimation of damage to materials in this case is restricted to damage to zinc, galvanised steel, limestone, sandstone, natural stone, mortar and rendering due to SO_2 and O_3 pollution. It therefore underestimates 'true' damages, as it is suggested that there may also be damages to other materials, due to both SO_2 and O_3 (see the methodology section). According to Rabl (1998) and Berg (1990) damages due to soiling and O_3 damages to rubber goods may be significant. The monetary damage of air pollution on cultural buildings has also not been assessed in this study. Hence, there seems to be scope for significant improvement in this area.

The impacts of air pollution of SO_2 and O_3 above the assumed natural background levels are presented in Table 8.26. The results are not shown for all possible combinations of natural background levels as this would create a complicated impact matrix and the influence of the chosen O_3 natural

Table 8.24 Crop prices used for the damage estimation

Receptor	Monetary value as given in the methodology section (Euro/t)	Domestic prices 1994[a] (Euro/t)	International prices 1991[b] (Euro/t)
Barley	5.4	152	69
Oats	5.6	142	69
Potato	8.2	124[c]	8.2
Rye	15.6	157	56
Sugar beet	4.8	53	4.8
Wheat	9.6	159	110
Rice	15.6	–	N.A.
Sunflower seed	23 456	N.A.	N.A.
Tobacco	3902	–	N.A.
Peas and beans	554	240[d]	554

Notes:
[a] Producer prices (CBS, 1998).
[b] FAO (1993).
[c] Production weighted average of all potatoes.
[d] Production weighted average of all peas and beans.
N.A. = Not analysed.
– = Not grown in the Netherlands in 1994.

background level on the results is only of the order of a few per cent. SO_2 is the dominant pollutant. Therefore, only results for combinations of low SO_2 with low O_3, mid SO_2 with mid O_3 and high SO_2 with high O_3 are presented.

The monetary damages related to these impacts are estimated using the monetary values presented in the methodology section, as depicted in Table 8.26. The subtotal damages due to air pollution of SO_2 and O_3 above the recommended values for the natural background concentrations (1 ppbV SO_2 and 20 ppbV O_3) are around 10 million Euro or 0.004 per cent of Dutch GDP in 1994.

Conclusions

The Dutch case study assessed the physical and monetary damages to human health, crops and materials caused by SO_2, PM_{10}, O_3, CO and heavy metal concentrations above selected natural background levels in the Netherlands. The results are presented in Table 8.27.

Health damage stands out as the major category of environmental damage in monetary terms. Cancer and inhalation diseases are also included in the table though negligible and virtually equal to zero when

Table 8.25 Damage to crops due to air pollutants in the Netherlands in 1994 with different crop prices

Receptor	Damage with prices as presented in the methodology section (million Euro p.a.)				Damage with domestic prices (million Euro p.a.)				Damage with international prices (million Euro p.a.)			
	O_3	SO_2			O_3	SO_2			O_3	SO_2		
		Background level				Background level				Background level		
	AOT40c	0 ppbV	1 ppbV	2 ppbV	AOT40c	0 ppbV	1 ppbV	2 ppbV	AOT40c	0 ppbV	1 ppbV	2 ppbV
Barley	0	−0.019	−0.030	−0.042	0	−0.53	−0.83	−1.2	0	−0.24	−0.38	−0.53
Oats	0.013	−1.9E-03	−3.3E-03	−4.9E-03	0.33	−0.049	−0.084	−0.12	0.16	−0.024	−0.041	−0.060
Potato	8.6	−0.80	−1.3	−1.9	130	−12	−20	−28	8.6	−0.80	−1.3	−1.9
Rye	0.052	−7.4E-03	−0.013	−0.018	0.53	−0.075	−0.13	−0.18	0.19	−0.027	−0.045	−0.066
Sugar beet	0.0	−0.49	−0.77	−1.1	0	−5.4	−8.5	−12	0.0	−0.49	−0.77	−1.1
Wheat	1.4	−0.14	−0.22	−0.30	23	−2.3	−3.6	−5.0	16	−1.6	−2.5	−3.5
Rice	0	0.0	0.0	0.0	0	0	0	0	0	0	0	0
Sunflower seed	N.A.	N.A.	N.A.	N.A.	N.A.	N.A.	N.A.	N.A.	N.A.	N.A.	N.A.	N.A.
Tobacco	0	0	0	0	0	0	0	0	0	0	0	0
Peas and beans	N.A.	0.55	0.36	0.16	N.A.	0.24	0.15	0.071	N.A.	0.55	0.36	0.16
	Damage in million Euro/y				Damage in million Euro/y				Damage in million Euro/y			
Subtotal excluding benefits	10.1	0.55	0.36	0.16	153.8	0.24	0.15	0.071	25	0.55	0.36	0.16

Subtotal including benefits	10.1	−0.91	−2.0	−3.2	153.8	−20.2	−32.5	−46.5	25	−2.6	−4.7	−6.9
	Damage in % of 1994 GDP					Damage in % of 1994 GDP				Damage in % of 1994 GDP		
Subtotal excluding benefits	3.9E−03	2.1E−04	1.4E−04	6.4E−05	6.0E−02	9.2E−05	6.0E−05	2.8E−05	9.7E−03	2.1E−04	1.4E−04	6.4E−05
Subtotal including benefits	3.9E-03	−3.5E-04	−7.6E-04	−1.2E-03	0.060	−7.8E-03	−0.013	−0.018	9.7E−03	−1.0E−03	−1.8E−03	−2.7E−03

Notes:
A negative number indicates a benefit from increased pollution levels.
N.A. = Not analysed.

Table 8.26 Damage costs to materials due to air pollutants in the Netherlands in 1994

Receptor		Damage (million Euro p.a.)		
		Background level		
	SO_2	0 ppbV	1 ppbV	2 ppbV
	O_3	16 ppbV	20 ppbV	24 ppbV
Zinc		0.11	−0.0064	−0.056
Galvanised steel		11	−0.66	−5.8
Limestone		0.27	0.20	0.12
Sandstone		0.18	0.13	0.079
Natural stone		0.19	0.10	0.017
Mortar		0	0	0
Rendering		1.9	1.4	0.85
Paint		12.0	8.7	5.4
Subtotal (million Euro p.a.)		25.4	9.8	0.61
Subtotal (% of 1994 GDP)		9.9E-03	3.8E-03	2.4E-04

compared to other impacts. The uncertainty around the health damage is large; it is the product of uncertainty on appropriate background levels, the exposure–response relationships, the pollutant measurement and geographic interpolation, and the valuation of the health impacts. The entity and direction of other possible errors in health effects estimation is not clear.

The estimate of damage to materials in this case study is restricted to damage to zinc, galvanised steel, limestone, sandstone, natural stone, mortar and rendering due to SO_2 pollution and O_3. It therefore underestimates 'true' damages, as it is suggested that both SO_2 and O_3 may also cause damages to other materials. The monetary damages of air pollution on cultural heritage building has also been neglected in this study. Hence there seems to be scope for significant improvements in this area.

Due to a lack of reliable exposure–response relationships the estimate of damage to crops is restricted to the following crops: wheat, barley, rye, oats, potato, sugar beet, beans and peas. The total crop damage is dominated by O_3 damage. As more reliable exposure–response functions become available there is scope for a more comprehensive damage assessment. In the crop damage case study later in this book (Chapter 10) a first attempt is made to take into account possible economic

Table 8.27 Summary of damages due to air pollution in the Netherlands in 1994[a]

Receptor		Damage costs (million Euro p.a.)							
		Pollutant						Subtotal	
		SO_2	PM_{10}	O_3	$AOT4_c$[b]	CO	Heavy metals	Million Euro p.a.	% 1994 GDP
Health	Acute and chronic morbidity	2	4900	51		3		5 000	1.9
	Cancer and inhalation diseases (As, Cd) (3% discount rate)					0	0	0	0
	Acute mortality (3% discount rate)	90	200	100				400	0.15
	Chronic mortality (3% discount rate)		4700					4700	1.8
Crops [c]		0			150			150	0.060
Materials[d]		10						10	3.8E−03
Subtotal (million Euro p.a.)		100	10 000	150	150	3	0	10 000	3.9
Subtotal (% 1994 GDP)		0.039	3.8	0.059	0.060	1.0E−03	0	3.9	

Notes:
[a] The recommended background concentrations used in this summary are: $PM_{10} = 10\ \mu g/m^3$, $O_3 = 20$ ppbV, AOT40c = 0 ppmh, $SO_2 = 1$ ppbV, CO = 0.15 ppbV and heavy metals = 0 ng/m3.
[b] Related to ozone concentration.
[c] Domestic prices for 1994; benefits from SO_2 fertilisation are not included.
[d] The damages due to ozone are included in the damages due to SO_2.

adjustments to air pollution by farmers and the effects of these adjustments on markets.

Although there is evidence for ecosystem and forest damage due to air pollution in the Netherlands, no widely applicable data on monetary valuation of these damages is readily available in the literature. This seems to be an area where progress is needed, but difficult to achieve. In fact, these impacts were not assessed in this study.

In summary, total air pollution damage in the Netherlands in 1994 was assessed to be of the order of magnitude of Euro 10 billion. This damage amounts to about 3.9 per cent of the Dutch gross domestic product (GDP) in 1994. The order of magnitude of the damage assessed in this study is to a large extent in accordance with a previous Dutch valuation study (Jansen, 1988) and with results from the previous phase of this research (Markandya and Pavan, 1999). It is important to note that damage from increased mortality, which is probably the most controversial part of a valuation exercise like this, accounts for half of this total damage. Furthermore, not including the 'chronic bronchitis' and 'restricted activity day' estimates, as these seem to overestimate the impacts, will reduce the damage costs by roughly Euro 4.7 billion to Euro 5.3 billion (2.1 per cent of Dutch GDP in 1994).

8.3.4 The United Kingdom

Human health damages

The results for the UK assessment are shown in the tables below, and provide results for health damages (acute and chronic effects on mortality and morbidity) for SO_2, ozone and PM_{10}.

Morbidity and mortality effects in money terms from SO_2 concentrations are shown in Tables 8.28 and 8.29, using the recommended exposure–response functions and monetary values for health endpoints, whereas Tables A.8.20 and A.8.21 in the Appendix display the number of cases occurring.

Cases of morbidity (acute and chronic) from PM_{10} concentrations are shown in the Appendix (Table A.8.22), using the recommended exposure–response functions. Cases of 'net' restricted activity days are also presented, taking account of other health endpoints, to avoid double counting. Monetary values of morbidity (acute and chronic) from PM_{10} concentrations are shown in Table 8.30. The net values for restricted activity days are used in the summary tables. There is particular uncertainty in the assessment of the incidence of chronic bronchitis.

The cases and monetary value of acute mortality from PM_{10} concentrations are shown in Table A.8.23 in the Appendix and Table 8.31 respectively, using the recommended exposure–response functions.

Table 8.28 Estimated monetary value for morbidity from SO_2 in the UK in 1996

SO_2 morbidity	Human morbidity (million Euro)			
	Background SO_2			
Health (morbidity)	Cost per unit (Euro)	0 ppbV	1 ppbV	2 ppbV
Resp. hosp. admissions in total population	7870	10	7	5

Table 8.29 Estimated monetary value for acute mortality from SO_2 in the UK in 1996

SO_2 acute mortality	Human mortality (million Euro p.a.)		
	Background SO_2		
Health (morbidity)	0 ppbV	1 ppbV	2 ppbV
Low 147 000 Euro (3%)*0.75	550	420	290
High 310 500 Euro (11%)*0.75	1200	880	600

The years of life lost (YOLL) and monetary value of chronic mortality from PM_{10} concentrations are shown in Table A.8.24 of the Appendix and Table 8.32 respectively. For the summary tables, acute and chronic mortality from PM_{10} are not thought to be additive and therefore in those tables we only report chronic values, to avoid double counting.

Monetary values of morbidity and mortality from ozone concentrations are shown in Tables 8.33 and 8.34, using the recommended exposure–response functions and monetary values for health endpoints. Cases of 'net' minor restricted activity days are also presented, taking into account other health endpoints, to avoid double counting. Cases of morbidity and mortality are in the Appendix (Tables A.8.25 and A.8.26).

For the reporting of mortality in the summary tables, we assume that acute and chronic mortality impacts from PM_{10} are not additive. However, chronic mortality impacts from PM_{10} and acute mortality impacts from ozone and SO_2 are additive. The values used in the summary tables are shown in Tables 8.35–8.36.

These results suggest that there are a total of 32 518 cases of mortality whose timing can be linked to short-term exposure to air pollution

Table 8.30 Estimated monetary value for morbidity from PM_{10} in the UK in 1996

PM_{10} morbidity	Cost per unit (Euro)	Human morbidity (million Euro p.a.) Background PM_{10} 0 μg/m³	5 μg/m³	10 μg/m³	15 μg/m³
Congest. heart failure in adults > 65 years	7870	27	21	14	8
Chronic bronchitis in adults	240 000	11000	8400	5800	3100
Restricted activity days in adults	75	1800	1300	920	500
NET restricted activity days in adults*	75	1700	1300	900	500
Bronchodilator usage in adult asthmatics	7.5	57	44	30	16
Cough days in adult asthmatics	7.5	59	45	31	17
Lower resp. symp. days in adult asthmatics	7.5	21	16	11	6
Bronchodilator usage in child asthmatics	7.5	14	11	7	4
Cough days in child asthmatics	7.5	24	18	12	7
Lower resp. sympt.days in child asthmatics	7.5	18	14	10	5
Case of chronic bronchitis in children	225	86	65	45	24
Chronic cough episode in children	225	110	84	58	31
Cerebrovascular hosp. adm. in total pop.	7870	46	35	24	13
Resp. hospital admissions in total pop.	7870	19	14	10	5

Note: * Restricted activity days (RAD) are also presented as net damage values in the Table. This assumes that all days in hospital for respiratory admissions (RHA), congestive heart failure (CHF) and cerebrovascular conditions (CVA) (assuming that the average stay for each is 10, 7 and 45 days respectively) are subtracted from the total restricted activity days (RAD).

Table 8.31 Estimated monetary value for acute mortality from PM_{10} in the UK in 1996

PM_{10} acute mortality	Human mortality (million Euro p.a.)			
	Background PM_{10}			
	0 μg/m^3	5 μg/m^3	10 μg/m^3	15 μg/m^3
Low 147 000 Euro (3%)*0.75	580	440	300	170
High 310 500 Euro (11%)*0.75	1200	940	650	350

Table 8.32 Estimated monetary value for chronic mortality from PM_{10} in the UK in 1996

PM_{10} chronic mortality*	Human mortality (million Euro p.a.)			
	Background PM_{10}			
	0 μg/m^3	5 μg/m^3	10 μg/m^3	15 μg/m^3
Low (11%)	5600	4200	2900	1600
High (3%)	13 000	10 000	7000	3800

Note: * Using values of 10 and 24 million Euro per one-year reduction of 21 μg/m^3 PM_{10} per 100 000 people. The values presented are the total damages calculated with the function, and so also include acute mortality damages.

(including PM_{10}, ozone and SO_2) in the UK. Accounting for recommended background concentrations, additional mortality corresponds to *13 944 cases*. The monetary value of these acute cases is about 3.6 to 7.6 billion Euro for total effects *and 1.5 to 3.2 billion Euro* adjusting for recommended background concentrations. Note that by accounting for background levels in this way we implicitly assume that all fine particles are equally aggressive to health. There are good grounds for regarding anthropogenic particles as likely to be more damaging than natural dusts, due to acidity, heavy metal content and so on, so the methods used could underestimate total damages linked to human activity.

The net value of total mortality from all pollutants (excluding acute PM_{10} mortality) is estimated at (low/high) about 12 to 16 billion Euro for total effects *and 5.5 to 8.2 billion Euro* adjusting for recommended background concentrations.

It is interesting to note that the ratio of impacts has changed since the previous phase of this project (Markandya and Pavan, 1999), due to the use

Table 8.33 Estimated monetary value for morbidity from O_3 in the UK in 1996

O_3 morbidity	Cost per unit (Euro)	Human morbidity (million Euro p.a.)			
		Background O_3			
		0 ppbV	16 ppbV	20 ppbV	24 ppbV
Minor restricted activity days in adults	75	1950	900	650	390
NET minor rest. activity days in adults*	75	1940	900	640	390
Asthma attacks in all asthmatics	37	190	86	61	37
Resp. hosp. adm. in total population	7870	93	43	31	18
Symptom days in total population	7.5	830	384	270	160

Note: * Minor restricted activity days (RAD) are also presented as net cases in the table. This assumes that all asthma attacks (AA) are also minor restricted activity days (MRAD) and that 3.5 per cent of the adult population (80 per cent of the total population) are asthmatic.

of European functions; these give higher acute mortality rates from ozone and lower acute mortality rates from PM_{10} compared to the previous US functions used.

The total values for morbidity in the UK are shown in Table 8.36. The total value of morbidity is estimated at 16 billion Euro for total effects and approximately 8 billion Euro adjusting for recommended background concentrations.

Crop damages

For the assessment of crop damages, a slightly different approach was taken, because of the availability of data. Crop production and land use data were not available on a fine resolution for the UK, and therefore the crop damage calculations have been implemented using the ALPHA model, which has been designed to assess damage on a European scale. This model is based on the European EMEP 150 km grid.

Table 8.34 Estimated monetary value for acute mortality from O_3 in the UK in 1996

O_3 acute mortality	Human mortality (million Euro)			
	Background O_3			
	0 ppbV	16 ppbV	20 ppbV	24 ppbV
Low 147 000 Euro (3%)*0.75	2500	1100	810	490
High 310 500 Euro (11%)*0.75	5200	2400	1700	1000

Table 8.35 Estimated monetary value for acute mortality in the UK in 1996

Mortality*	Human mortality (million Euro p.a.)				
	Background	No background		Background	
	Discount rate	3%	11%	3%	11%
Acute mortality SO_2	(1 ppbV)	550	1200	420	880
Acute mortality PM_{10}*	(10 $\mu g/m^3$)	580	1200	300	650
Acute mortality O_3	(20 ppbV)	2500	5200	810	1700
Chronic mortality PM_{10}	(10 $\mu g/m^3$)	13000	5600	7000	2900

Note: * Acute and chronic mortality from PM_{10} are not additive and so only the values for chronic mortality are reported in the summary table. Also, the 3 per cent discount rate produces a higher value for chronic mortality, but a lower value for acute mortality.

Table 8.36 Estimated monetary value for morbidity effects in the UK in 1996

Morbidity		Human morbidity (million Euro p.a.)	
	Background	0 $\mu g/m^3$	Background
SO_2	(1 ppbV)	10	7
PM_{10}	(10 $\mu g/m^3$)	13000	6900
O_3	(20 ppbV)	3000	1000

The ozone and SO_2 data required for the crop damage calculations were converted to the EMEP grid by estimation from the 1×1 km maps described in an earlier section. The damages from crop impacts in the UK are shown in Table 8.37. The total damage to UK crops is estimated as 750 million Euro.

Table 8.37 Estimated monetary damage from effects of SO_2 and O_3 on crops in the UK in 1996

	Million Euro p.a.	
	O_3 (AOT40)	SO_2
Background	Zero	1 ppbV
Arable crops	680	70

Table 8.38 Estimated monetary damage from effects of SO_2 and acidity on material in the UK in 1996

Material	Material loss (million Euro p.a.)	
	Background	
	SO_2	Acidity
	1 ppbV	pH 5
Natural stone	9	2
Mortar	15	2
Paint (carbonate)	460	42
Galvanised steel	670	59

Material damages

Monetary damages from material impacts in the UK are shown in Table 8.38, whereas physical impacts are illustrated in the Appendix to this chapter (Table A.8.27). The analysis has used the four-year exposure–response functions from the UNECE Integrated Collaborative Programme (Kucera, 1994), as the eight-year ICP functions were not available in time for the study. The function for carbonate paint (Haynie, 1986) was also used. Only monetary damages are provided for this stock at risk.

The total damage to UK materials is estimated as being approximately 1.3 billion Euro. This value does not include building soiling from particulates and effects of ozone on rubber materials.

Conclusions

The results from the previous sections are brought together in Table 8.39. The results show that PM_{10} damages are responsible for most damage (76 per cent), with lesser effects from ozone (14 per cent) and SO_2 (9 per cent). In terms of impact categories, mortality (44 per cent) and morbidity

Table 8.39 Summary table of damages from air pollution in the UK

Impact category	Damage costs (million Euro)				
	PM_{10}	Ozone	SO_2	Acidity	Total
Morbidity	7000	1000	8	–	8000
Mortality (3% discount rate (dr))	6700*	810	420	–	8000
Material	NQ	NQ	1150	110	1300
Crops	–	680	70	–	750
Total damage costs	14000	2500	1600	110	18000

Notes: * Chronic PM_{10} damages only to avoid double counting. NQ: not quantified. Background rates in the summary table assume: SO_2: 1 ppbV; PM_{10}: 10 $\mu g/m^3$; O_3: 20 ppbV.

(44 per cent) dominate damages. The damages from materials and crops impacts represent 7 and 4 per cent respectively.

8.3.5 Summary of Results – All Countries

Table 8.40 shows damage costs estimated for the countries of study (Germany, Italy, the Netherlands and the United Kingdom), ordered by the impact categories mortality, morbidity, crops, and materials. Base years for the calculations are 1994 for Germany, Italy and the Netherlands and 1996 for the UK.

The pollutants which were taken into account were SO_2, PM_{10} and O_3 except for a few exceptions. These were: no calculations for O_3 and material damages in Italy; additional calculations of CO morbidity impacts (2.6 million Euro p.a.) and the assumption of a constant level for wet acid deposition for the Netherlands; damage estimations due to acidity in the UK (about 110 million Euro p.a.). These exceptions and the different base years have to be taken into account in comparing the various estimates. Also additional effects like soiling caused by PM_{10} and damages on rubber and paints due to ozone which contain large uncertainties are not included in the core analysis. These impacts are discussed in the sensitivity analysis section (Section 8.4). In addition, not all environmental impacts of the pollutants can be expressed in exposure–response functions, because in some areas, for example, natural ecosystems, the processes are very complex and the impacts of pollutants are not yet well known. Concerning these effects, the exceeding of critical levels could only be assessed for some pollutants.

The values estimated in total were for Germany and for Italy about 45 billion Euro p.a., for the Netherlands 10 billion Euro p.a. and for the

Table 8.40 Damage costs caused by the pollutants SO_2, PM_{10} and O_3 in Germany, Italy, the Netherlands and the UK

	Germany	Italy[a]	Netherlands[b]	United Kingdom[c]
	Damage costs (million Euro p.a.) unless otherwise stated			
Human health				
Mortality	22 000	23 000	5 000	8 000
Morbidity	21 000	22 000	5 000	8 000
Subtotal	43 000	45 000	10 000	16 000
Percentage of GDP (%)[d]	2.7	4.4	3.9	1.8
Costs per inhabitant (Euro/person p.a.)	530	780	650	270
Crops				
Subtotal	1600	2	150	750
Percentage of GDP (%)	0.10	2E −4	0.06	0.08
Costs per inhabitant (Euro/person p.a.)	20	4E − 2	10	13
Material				
Subtotal	130	N.A.	10	1300
Percentage of GDP (%)	0.01	N.A.	4 e-3	0.14
Costs per inhabitant (Euro/person p.a.)	2	N.A.	1	21
Total	45 000	45 000	10 000	18 000
Percentage of GDP (%)	2.8	4.4	3.9	2.0
Costs per inhabitant (Euro/person p.a.)	550	780	660	300

Notes:
[a] Results for Italy do not include damages due to O_3 and material impacts.
[b] Results for the Netherlands include morbidity impacts of CO at 1.8 million Euro/p.a. (assumed background for CO: 0.15 ppbV); wet acid deposition was assumed to be the average value of 100 meq/m^{2}a p.a..
[c] Results for the UK include material damages due to acidity.
[d] European Commission (1997a); Eurostat (1997).
The background levels considered were PM_{10}: 10 μg/m^3, O_3: 20 ppbV (AOT40 crops: 0 ppbVh), SO_2: 1ppbV.

United Kingdom 18 billion Euro p.a. In addition to the absolute damage costs, other indicators are presented: damage costs per inhabitant in one year (unit: Euro per person and year) and damage cost per GDP (unit: percentage of GDP). The highest damage costs per inhabitant were observed for Italy at 780 Euro per person in one year which also shows the highest value of GDP-referred costs at 4.4 per cent, although O_3 was not considered for the assessment. The Netherlands at 660 Euro per person in one year and 3.9 per cent of GDP, Germany at 550 Euro per person in one year and 2.8 per cent of GDP, and finally the UK at 300 Euro per person in one year and 2.0 per cent of GDP follow.

The health damages represent by far the largest share of damage costs in each country. The major contribution is estimated for three impacts caused by PM_{10}, namely chronic mortality, chronic bronchitis and restricted activity days. This can be seen especially in the damage costs in Italy, for which the highest damage costs to human health are observed at 45 billion Euro p.a., although impacts due to O_3 are not included in the Italian analysis. Estimates of O_3 related health damage costs for the remaining countries (background level: 20 ppbV) result in values of approximately the same order of magnitude as the damage costs for crops and material (Germany: 3100 million Euro p.a., the Netherlands: 150 million Euro p.a. and the UK: 1800 million Euro p.a.).

Ozone is the major contributor to damages to field crops for Germany, the Netherlands and the UK. As no O_3 was taken into account for the Italian study, the amount of damage costs assessed for crops in Italy is very low. Concerning material damages the greatest damage costs were observed for the UK. These are dominated by effects caused by SO_2.

8.4 SENSITIVITY ANALYSIS

8.4.1 Introduction

A sensitivity analysis has been undertaken for the UK to investigate a number of additional issues for air pollution. The sensitivity assessment comprises three parts:

1. a study of the year on year changes in pollution levels and damages;
2. the assessment of additional impacts, which cannot be assessed by all teams because of data limitations; and
3. the use of alternative exposure–response functions for health effects.

A preliminary sensitivity analysis for the Netherlands is also reported.

8.4.2 Year on Year Changes

As well as the annual damages presented in the main part of the report, it is also interesting to look at changes over time. This reveals the effect

Table 8.41 UK atmospheric emissions 1990–1996

	ktonnes of pollutant						
	1990	1991	1992	1993	1994	1995	1996
PM_{10}*	314	313	299	293	267	220	213
SO_2	3764	3579	3477	3166	2705	2351	2028
NO_x	2752	2653	2572	2408	2297	2145	2060
NMVOC	2632	2578	2489	2375	2331	2200	2111

Note: * The values for PM_{10} only include primary emissions. Estimates of secondary pollutants (for example, sulphates and nitrates) that also contribute to ambient PM_{10} concentrations are not included.

Source: DETR (1998).

of recent legislation. Year on year change is also more appropriate for comparison against the traditional economic indicators used in national accounting.

Ideally, such an analysis would be undertaken using pollution maps from previous years. There is however a problem with this. The methods and levels of resolution of pollution maps have improved significantly over recent years in the UK. Therefore, any differences in damages between years are likely to be due to changes in methodology, rather than genuine changes in pollution levels.

Instead, we have used a simpler approach, based on national emissions forecasts over time. For these data, estimates from previous years are also updated with methodological advances, and the data should represent accurate relative changes in pollution levels. However, there is one disadvantage to this approach. Within the green accounting framework, we are interested in ambient pollution levels. Ambient levels in the UK are the result of both UK and European emissions (domestic and trans-boundary pollution). The approach below assumes that the concentration profiles over time are directly related to UK emissions, that is, we assume that the contribution from European emissions to UK ambient concentration levels would follow a similar profile. The data for UK emissions is shown in Table 8.41.

In order to look at the change in damages between years, we must be able to convert pollutant concentrations to emissions. We have therefore derived unit pollution costs from the 1996 pollution maps and the 1996 emissions in Table 8.42. It should be noted that the values would be different to unit pollution costs derived in the ExternE Project, as the latter include secondary particulates in the analysis of PM_{10}.

Table 8.42 Scaling factors for major pollutants, based on UK damages

	1996 (ktonnes)	Total UK damage (million Euro)	Unit cost (Euro/t)
PM_{10}#	213	13 516	63 455
SO_2	2028	1886	930
NO_x*	2060		
NMVOC*	2111		
Ozone*	4171	2777	666

Notes:
Only primary emissions of PM_{10} are included. We have assumed that changes in primary particulates will be representative of both primary and secondary particulates (that is, total PM_{10}). We have not separately assessed the contribution of NOx and SO_2 to PM_{10}.
* For ozone, we have simply assumed NOx and NMVOC precursors make an equal contribution to ozone formation. This is consistent with the recommendations of the ExternE Core project on unit pollution costs for ozone.

Table 8.43 Total estimated UK damages 1990–1996

	Million Euro						
	1990	1991	1992	1993	1994	1995	1996
PM_{10}	19 925	19 862	18 973	18 592	16 943	13 960	13 516
SO_2	3500	3328	3234	2944	2516	2186	1886
Ozone	3586	3484	3371	3185	3082	2894	2778
Total	27 011	26 674	25 577	24 722	22 540	19 040	18 180

These factors are used to analyse the emissions from previous years and so calculate associated damages. The results are shown in Table 8.43.

The results show the year on year reductions in total damages from the 1990s onwards. This is due to large reductions in SO_2 and PM_{10} levels, both of which are largely due to the reduction in coal combustion in the UK.

8.4.3 Extension to Other Pollutants

There are a number of other pollutants which have not been assessed in the main analysis, but which are suspected to cause health impacts. There are also a number of additional non-health impacts from the main pollutants assessed in the main analysis. The reasons for excluding these pollutants include a lack of measurement data, but in addition the damages from these pollutants are expected to be low compared to the major impact pathways for PM_{10}, SO_2 and ozone – their omission is therefore unlikely to affect the overall results.

The sensitivity analysis estimates the level of damages of these additional pollutants, and provides information on the relative importance of each. The analysis assesses:

- health effects of carbon monoxide, benzene, 1.3-butadiene, lead and other heavy metals;
- material damage from soiling and from ozone.

Sensitivity analysis has been undertaken for the UK. However, some analysis was included for the Netherlands, as shown in Tables 8.20 and A.8.15, for some selected heavy metals (arsenic and cadmium). Evidence shows that the cancer impacts caused by inhalation of these heavy metals are negligible.

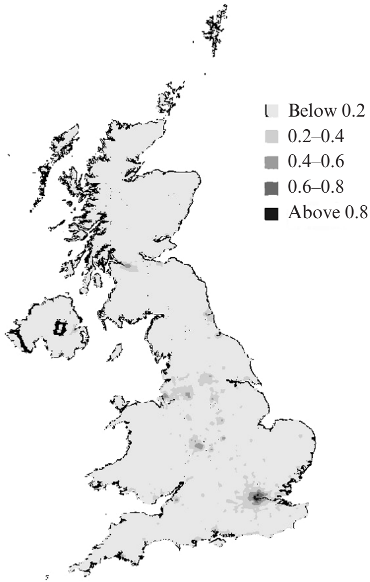
Air quality data

Air pollution data covering the whole UK for carbon monoxide, lead, benzene and 1.3-butadiene were provided by Stedman (1998) at a resolution of 1×1 km.

Carbon monoxide (CO) The map of carbon monoxide (Figure 8. 15) was derived from combining a rural map (assumed to be a constant level of 0.15 ppm) and the relationship between measured ambient concentrations in urban areas and CO emissions from vehicles.

Benzene and 1.3-butadiene The maps of benzene and 1.3-butadiene (Figures 8.16 and 8.17) were derived directly from VOC emission inventories in combination with the interpolated map of rural NO_2 levels. Benzene and 1.3-butadiene were assumed to be 1.55 and 0.23 per cent of total VOC emissions respectively. Emissions are mostly associated with vehicles, and are highest in areas of slow-moving traffic.

Lead Annual mean concentrations of lead are available for ten background monitoring sites in the UK. In most areas, airborne lead concentrations are dominated by the contribution from vehicles, owing to the continued use of leaded petrol. It is not possible to model the contribution of industrial lead sources, and therefore the map used in this study (Figure 8.18) represents the transport derived background lead concentration. The map is based on the relationship between lead concentrations and local NO_x from vehicles.



Note: This map has been plotted at a resolution of 1 × 1 km.

Figure 8.15 Map of annual mean CO levels (ppm) for the UK

Table 8.44 Background levels of pollutants used in the assessment

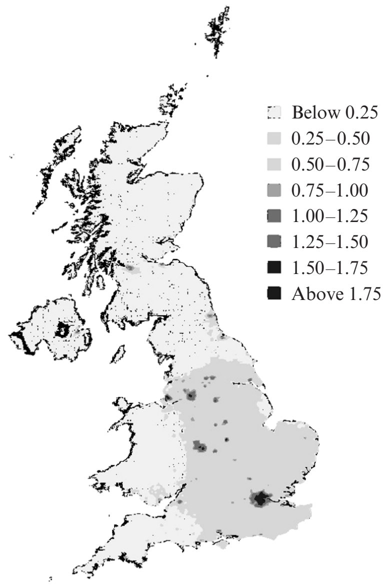
Pollutant	Background
Lead	0
Benzene	0
1.3-butadiene	0
CO	0.15 ppmV

Background concentrations

In addition to the description of the data sources, background (natural) levels of each pollutant have been estimated. These are shown in Table 8.44.

Results for the United Kingdom

The methodology and results for the UK assessment are shown in Tables 8.45 to 8.52.



Note: This map has been plotted at a resolution of 1 × 1 km.

Figure 8.16 Map of annual mean benzene levels (ppb) for the UK

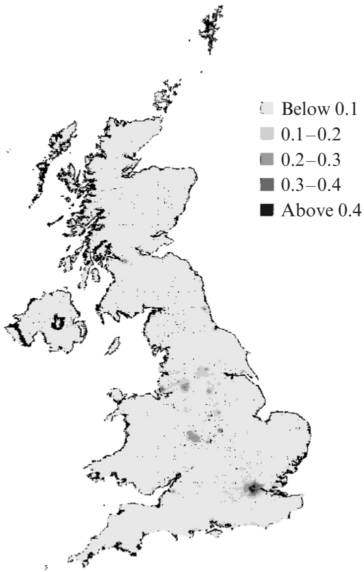
Carbon monoxide The recommended exposure-response function for effects of CO on morbidity is shown in Tables 8.45 and 8.46.

The damage estimates of morbidity effects from CO concentrations in the UK are shown below. In the Appendix to this chapter the number of cases are displayed (Table A.8.28).

Benzene and 1.3-butadiene For benzene and 1.3-butadiene, the estimated cancer risk has been quantified using unit risk factors (URFs) from the literature. These are the estimated probabilities that a person of standard weight will develop cancer due to exposure (by inhalation) to a concentration of $1 \mu g/m^3$ of pollutant over a 70-year lifetime. For the purposes of our calculations we assume the functions to be linear and assumed a one-year increment in exposure to increase the lifetime risk of cancer by 1/70th of the URF.

The valuation of cancer is treated in a similar way to chronic mortality. The assumptions for valuation are set out in Table 8.47. Note the cost of illness (COI: 250 000 Euro) is added, because the period of illness prior to death is not adequately taken into account in the VOLY concept.

For this calculation, we assume all cancers are fatal, though this will overestimate damages.



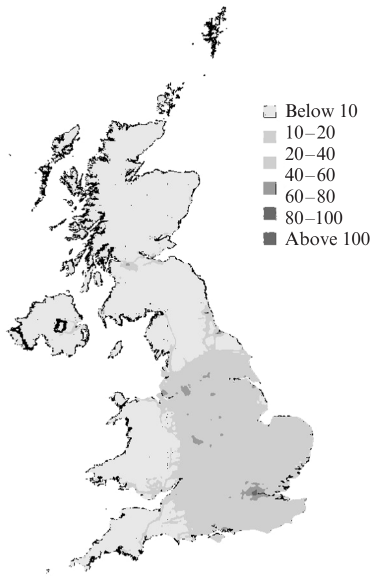
Note: This map has been plotted at a resolution of 1×1 km.

Figure 8.17 Map of annual mean 1.3-butadiene levels (ppb) for the UK

The damage estimates of cancer cases from benzene and 1.3-butadiene concentrations are shown in Table 8.48, using the recommended exposure–response functions and monetary valuation of endpoints. Again, the number of cases arising from exposure to benzene and 1.3-butadiene are presented in the Appendix (Table A.8.29).

Lead The recommended exposure–response functions for effects of lead on morbidity is: 2.57 IQ points per 10 mg/dl blood level, where IQ = intelligence quotient. The function only applies to younger children, as adverse effects of lead exposure is greatest when the child's nervous system is developing rapidly. For this study, we have assumed that only children aged 0–2 are affected. The conversion from lead air concentrations to lead blood levels assumes that 1 mg/m^3 in air is equivalent to 5 mg/dl in blood. At present no recommended valuation is available for IQ points and so no estimates of damages have been made.

For comparison, the levels of lead emissions in 1995 were estimated at 1468 tonnes (Salway et al., 1997). This compares to 2703 tonnes in 1990 and 8051 tonnes in 1980. Therefore, current damages from lead pollution to health should be around 20 per cent of the values when leaded fuel was



Note: This map has been plotted at a resolution of 1 × 1 km.

Figure 8.18 Map of annual mean lead levels (ng/m^3) for the UK

widely used. These levels will decrease further as the use of leaded petrol declines, and is ultimately phased out.

Other heavy metals A simplified approach has been used to look at the potential impacts from other heavy metals in the UK. Data have been taken on total atmospheric emissions of metals (Salway et al., 1997), as air pollution maps for these metals are not currently available.

These emission data have been combined with previous estimates of unit pollution costs for each metal (that is, damage costs per tonne of pollutant released) to calculate the potential damages from 1995 emissions. The unit pollution costs are taken from previous ExternE studies which have used the recommended unit risk factors (URFs) specified in the ExternE Core Project. Only impacts from direct inhalation are quantified. No account is taken of other potential routes of exposure, such as through deposition and wind-blown dust or via the food chain. Only local scale effects are included (that is, only local scale dispersion models are used).

The assumptions in valuation (with lung cancer as the endpoint) are set out in Table 8.50.

The resulting damages for 1995 are shown in Table 8.51.

Table 8.45 Exposure–response function for quantification of human health impacts from CO

Receptor	Impact category	Reference	f_{er}[1]	Uncertainty rating
Elderly 65+	Congestive heart failure	Schwartz and Morris, 1995	5.55E-7	B

Note:
[1] The exposure–response slope, f_{er}, is for Western Europe and has units of cases (yr-person-µg/m³).

Table 8.46 Estimated monetary value for morbidity from CO in the UK in 1996

CO morbidity	Human morbidity (million Euro)	
	Background CO	
	0 ppmV	0.15 ppmV
Congest. heart failure in adults > 65 years 7870 Euro per case	12.50	5.77

Table 8. 47 Assumptions for valuation of cancer mortality cases

Cancer types	Leukaemia
Causative pollutants	benzene, butadiene
Latency l in years	8
YOLL	22
Discount rate: r = 3%; VLYL(r = 3%) = 155 000	
Value of a fatal case	1 813 030
COI	250 000
Total (rounded)	2 060 000
Discount rate: r = 10%; VLYL(r = 10%) = 312 000	
Value of a fatal case	1 175 770
COI	250 000
Total (rounded)	1 430 000

Table 8.48 Estimated monetary value for cancer risk from benzene and 1.3-butadiene in the UK in 1996 (sensitivity)

Cancer risk	Human morbidity (million Euro)	
	Background 0 μg/m³	
	10% dr 1 430 000 Euro	3% dr 2 060 000 Euro
Benzene	20.49	29.52
1.3-butadiene	99.89	143.90

Table 8.49 Estimated morbidity impacts resulting from lead in the UK in 1996 (sensitivity)

Lead morbidity	Human morbidity (IQ points)
	Background 0 ppmV
Total loss of child IQ points < 2 years	69 800

Table 8.50 Assumptions made in valuation

Cancer types	Lung cancer
Causative pollutants	Metals
Latency l in years	15
YOLL	16
Discount rate: r = 3%; VLYL (r = 3%) = 155 000	
Value of a fatal case	1 080 000
COI	250 000
Total	1 330 000
Discount rate: r = 10%; VLYL (r = 10%) = 312 000	
Value of a fatal case	420 000
COI	250 000
Total	670 000

Impacts from inhalation of heavy metals, as discussed for the Netherlands, are negligible when compared to the other impacts considered in the main analysis.

Other material damages There are two additional impact pathways for materials. The first is building soiling from particulates. Building soiling

Table 8.51 1995 UK atmospheric emissions, impacts and damages from heavy metals (sensitivity)

Metal	Emission (tonnes)	URF $\mu g/m^3$ / yr	Unit damage (Euro/t)		Total (million Euro)	
			3%	10%	3%	10%
Cadmium	24	2.5×10^{-5}	3195	1610	0.077	0.039
Mercury*	20		–			–
Nickel	402	5.6×10^{-6}	710	357	0.285	0.144
Chromium	60	5.6×10^{-4}	71 001	35 767	4.260	2.146
Arsenic	98	4×10^{-5}	5333	2687	0.523	0.263

Note: * Mercury is not carcinogenic to humans. A threshold for non-carcinogenic effects has been proposed as $1 \mu g/m^3$ for inhalation effects. Ambient levels are generally very low (that is, <10 ng/m^3).

damages were calculated using the approach by Pons et al. (1995). This suggested the following function for quantifying building soiling:

$$S_i = a * P_i * \Delta TSP_i \quad \text{(and } a = b * 2)$$

where:

S_i = Annual soiling damage at receptor location i.
P_i = Number of people in location i.
ΔTSP_i = Change in annual average TSP in $\mu g/m^3$.
a = WTP per person per year to avoid soiling damage of $1 \mu g/m^3$ particles.
b = Cleaning costs per person per year from a concentration of 1 $\mu g/m^3$ of particles.

A value for b of 0.5 Euro has been suggested by Pons et al. (1995), based on cleaning costs data from Paris.

Using this function, the estimated total damages for the UK (based on the PM_{10} map) are 1170 million Euro, with estimates of 892 million, 613 million and 335 million Euro for background concentrations of 5, 10 and 15 $\mu g/m^3$ respectively.

There is some uncertainty with these values. The cleaning costs are taken from Paris and it is not known if there is a threshold. Studies in the UK indicate that the size of the UK market for stone cleaning is around 150 million Euro per year. Therefore, the above values, which are based on WTP and include amenity impacts, do not seem unrealistic.

Table 8.52 Summary table of additional damages to the core analysis from air pollution in the UK (sensitivity)

Impact category	Damage costs (million Euro)					
	PM_{10}	Ozone	CO	Benzene/ Butadiene	Heavy metals	Total
Materials	340	120				460
Morbidity	–	–	6			6
Mortality (3% dr)	–	–		170	5	180
Total damage costs	340	120	6	170	5	640

Notes:
Backgrounds assumed: PM_{10}: 10 $\mu g/m^3$; CO: 0.15 ppmV, and zero for O_3, benzene, butadiene, heavy metals.

The second additional impact pathway for material damage is for ozone. A recent study estimates that ozone damage to paint is likely to be insignificant, though damage to rubber goods could be 50 million to 265 million Euro per year with a best estimate of 120 million Euro (Holland et al., 1998).

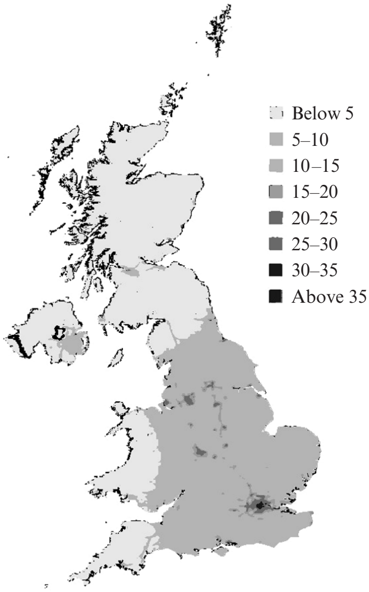
Summary
The results from the previous sections are brought together in Table 8.52. The results show that additional material damages are significant. Damages from SO_2 and acidity only totalled 1300 million Euro in the UK (see Table 8.39). Added to the damages in Table 8.52, this gives total material damages of around 1800 million Euro.

The additional health damages are less important. Morbidity and mortality damages from the other pollutants total more than 180 million Euro in the UK, but this is low when compared to the health effects from PM_{10}, SO_2 and ozone (which had total morbidity impacts of 7932 million Euro and total mortality impacts of 8242 million Euro in the UK).

8.4.4 Use of Alternative Exposure–Response Functions

A number of additional exposure–response functions were identified in the ExternE Core Project, which, while excluded in the best estimates, were recommended for use in sensitivity analysis. A number of types of additional functions have been assessed.

The first group include functions that attribute major health effects to other pollutants, such as NO_2 and CO, rather than particulates. For example, exposure–response functions from the EC APHEA study (Katsouyanni et al., 1997) somewhat strengthen the possible relationship between NO_2 and



Note: This map has been plotted at a resolution of 1 × 1 km.

Figure 8.19 Map of the annual mean NO_2 levels (ppb) for the UK

acute mortality, with some positive relationships being shown in several European cities.

The second group include relationships from the North American literature which have been excluded on the basis of newer European functions or because the endpoints they quantify are not relevant in the European context (for example, emergency room visits – ERVs).

Finally, this section also includes an assessment of damages from $PM_{2.5}$ as an alternative metric for quantification of particulates by use of PM_{10}.

Air quality data

Air quality data covering the whole UK for nitrogen dioxide and $PM_{2.5}$ were provided by Stedman (1998).

Nitrogen dioxide (NO_2) The NO_2 map for 1996 (Figure 8.19) has been produced using estimates of low-level emissions to calculate ambient concentrations. Urban background concentrations are determined to a large extent by NOx emissions from road transport, and there is a good correlation between annual mean urban background NO_2 concentrations and local emissions estimates. The urban background map was combined with a rural map, interpolated from monthly measurements of NO_2 by diffusion tubes.

$PM_{2.5}$ Ambient $PM_{2.5}$ is comprised of primary and secondary particulates. The concentrations of $PM_{2.5}$ have been derived from measurements of secondary particulates, and of NO_x. For the secondary particulates, it was assumed that 80 per cent of the secondary particulates within the PM_{10} range would be less than 2.5 μm. Primary particulates were interpolated using the NO_x map, on the basis of the relationship between $PM_{2.5}$ and NO_x as measured in Birmingham (Stedman, personal communication).

Background concentrations

In addition to the description of the data sources, background (natural) levels of each pollutant have also been estimated. For NO_2, the background concentration is set at 2 ppb.

The assessment of background $PM_{2.5}$ levels is more difficult. For background PM_{10} concentrations, it was assumed around 30 per cent were of natural origin. For background $PM_{2.5}$ levels, we have scaled down the PM_{10} estimates and used background values of 3, 6 and 9 $\mu g/m^3$ $PM_{2.5}$.

However, more recent reviews (QUARG, 1996) indicate that within the fine fraction (that is, $PM_{2.5}$) the levels of natural minerals may only be around 5 per cent (rather than the 20–25 per cent typically found in the total particulate mix). It may therefore be that natural levels of $PM_{2.5}$ are very low indeed – perhaps only 5 per cent of total measured values.

Results for the UK

The functions used in the sensitivity analysis for health effects are shown in Table 8.53 below.

The damage estimates of morbidity effects from PM_{10} concentrations from the use of these functions are shown in Table 8.54, whereas the cases are in the Appendix in Table A.8.30.

The damage estimates of morbidity effects from ozone concentrations from the use of the functions are shown in Table 8.55 below and the cases are in the Appendix in Table A.8.31.

The damage estimates of morbidity and mortality effects from NO_2 concentrations from the use of these functions are shown in Tables 8.56–8.57 (number of cases in the Appendix, Tables A.8.32–A.8.33).

The damage estimates of morbidity and mortality effects from CO concentrations from the use of these functions are shown in Tables 8.58 and 8.59 below. The corresponding case estimates are in the Appendix (Tables A.8.34–A.8.35).

Table 8.53 Human health E-R functions for sensitivity analysis (Western Europe)

Receptor	Impact category	Reference	Pollutant	f_{er}	Uncertainty
Elderly, 65+					
	Ischaemic heart disease	Schwartz and Morris, 1995	PM_{10}	1.75E-5	B
			CO	4.17E-7	B
Entire population					
	Respiratory hospital admissions	Ponce de Leon, 1996	NO_2	1.40E-6	A?
	ERV for COPD	Sunyer et al., 1993	PM_{10}	7.20E-6	B?
	ERV for asthma	Schwartz, 1993;	PM_{10}	6.45E-6	B?
		Bates et al., 1990			B?
		Cody et al., 1992	O_3	1.32E-5	
	ERV for croup in pre-school children	Schwartz et al., 1991	PM_{10}	2.91E-5	B?
	Acute mortality	Touloumi et al., 1994	CO	0.0015%	B?
		Sunyer et al., 1996, Anderson et al., 1996	NO_2	0.034%	B?

Notes: The exposure–response function is for Western Europe and has units of cases/(yr-person-$\mu g/m^3$) for morbidity, and the percentage change in annual mortality rate/($\mu g/m^3$) for mortality.

Sources: European Commission (1995) and Hurley and Donan (1997b).

The summary results of these sensitivity analyses are shown in Table 8.60 below.

It is also interesting to compare the mortality analysis in terms of cases for NO_2, SO_2, PM_{10}, O_3 and CO as well as YOLL shown in Table 8.62. The 'no background' analysis shows NO_2 and SO_2 give rise to an almost identical number of cases of mortality as PM_{10} (5689 and 5003 cases versus 5288 cases). The values for CO are slightly lower, but not very different (3116 cases). However, results change if we consider results obtained using recommended background levels.

Table 8.54 Estimated monetary value for morbidity from PM_{10} in the UK in 1996 (sensitivity)

PM_{10} morbidity		Human morbidity (million Euro)			
		Background PM_{10}			
	Cost per unit (Euro)	0 μg/m³	5 μg/m³	10 μg/m³	15 μg/m³
Ischaemic heart disease	7870	25.56	19.46	13.36	7.26
ERV for COPD	223	1.88	1.43	0.98	0.54
ERV for asthma	223	1.68	1.28	0.88	0.48
ERV for croup in pre-school children	223	7.59	5.79	3.98	2.17

Table 8.55 Estimated monetary value for morbidity from O_3 in the UK in 1996 (sensitivity)

O_3 morbidity	Human morbidity (million Euro)			
	Background O_3			
	0 ppbV	16 ppbV	20 ppbV	24 ppbV
ERV for asthma 223 Euro	9.81	4.57	8.17	7.35

Table 8.56 Estimated monetary value for morbidity from NO_2 in the UK in 1996 (sensitivity)

NO_2 morbidity	Human morbidity (million Euro)	
	Background NO_2	
	0 ppmV	2 ppmV
Respiratory hospital admissions 7870 Euro	16.31	13.97

For completeness, we have also presented an alternative function for chronic mortality. The original E-R functions from Pope et al. (1995) are expressed in terms of $PM_{2.5}$ or sulphates. There are therefore two possible ways of converting these functions to PM_{10}. Using the standard conversion

Table 8.57 Estimated monetary value for acute mortality from NO_2 in the UK in 1996 (sensitivity)

NO_2 acute mortality	Human mortality (million Euro)	
	Background NO_2	
	0 ppmV	2 ppmV
Low 147 000 Euro (3%)*0.75	627	613
High 310 500 Euro (11%)*0.75	1325	1294

Table 8.58 Estimated monetary value for morbidity from CO in the UK in 1996 (sensitivity)

CO morbidity	Human morbidity (million Euro)	
	Background CO	
	0 ppmV	0.15 ppmV
Ischaemic heart failure in adults > 65 years 7870 Euro	9.39	4.34

Table 8.59 Estimated monetary value for acute mortality from CO in the UK in 1996 (Sensitivity)

CO acute mortality	Human mortality (million Euro)	
	Background CO	
	0 ppmV	0.15 ppmV
Low 147 000 Euro (3%)*0.75	344	162
High 310 500 Euro (11%)*0.75	726	343

factors produces functions which differ by a factor of about 2. For the central estimates used by all teams in the main section, we have used the conversion from sulphates, as these give the lower results (and this is consistent with the European risk factors for acute mortality and PM_{10}, which are about a factor of 2 lower than the US studies). However, we have also used

Table 8.60 Estimated monetary value for morbidity in the UK in 1996 (sensitivity)

Pollutant	Damage costs (million Euro)	
	No background	background
PM_{10} morbidity	36.71	19.2
Ozone morbidity	9.81	8.17
NO_2 morbidity	16.31	13.97
CO morbidity	9.39	4.34

Table 8.61 Estimated monetary value for acute mortality in the UK in 1996 (sensitivity)

Mortality		Human mortality (million Euro)			
		No background		Background	
	Discount rate	3%	11%	3%	11%
Acute mortality NO_2	2 ppbV	627	1,325	613	1,294
Acute mortality CO	0.15 ppmV	344	726	162	343

Table 8.62 Estimated acute and chronic mortality impacts in the UK in 1996 (sensitivity)

Mortality		Human mortality (cases)	
	Background	No background	Background
Acute mortality NO_2	2 ppmV	5689	5559
Acute mortality SO_2	1 ppbV	5003	3798
Acute mortality PM_{10}	10 $\mu g/m^3$	5288	2771
Acute mortality O_3	20 ppbV	22 227	7375
Acute mortality CO	0.15 ppmV	3116	1473
		Human mortality YOLL	
Chronic mortality PM_{10}	10 $\mu g/m^3$	261 861	137 221

the function converted from $PM_{2.5}$ to see the impact on damages. The impacts and values from this analysis are presented in Tables 8.63 and 8.64. As can be seen from the results, this alternative conversion increases chronic mortality impacts and values significantly.

Table 8.63 Chronic mortality impacts resulting from PM_{10} in the UK in 1996 (sensitivity)

PM_{10} chronic mortality	Human mortality (cases)			
	Background PM_{10}			
	0 μg/m³	5 μg/m³	10 μg/m³	15 μg/m³
Chronic mortality Pope et al., 1995	549 909	419 037	288 165	157 293

Table 8.64 Monetary value for chronic mortality from PM_{10} in the UK in 1996 (sensitivity)

PM_{10} chronic mortality*	Human mortality (million Euro)			
	Background PM_{10}			
	0 μg/m³	5 μg/m³	10 μg/m³	15 μg/m³
Low (11%)	11 700	8 916	6 131	3 347
High (3%)	28 080	21 398	14 715	8 032

Notes:
* Using values of 10 and 24 million Euro per one-year reduction of 10μg/m³ PM_{10} per 100 000 people. The values presented are the total damages calculated with the function, and so also include acute mortality damages.
The exposure–response function, is for Western Europe and has units of cases/(yr-person-μg/m³) for morbidity, and the percentage change in annual mortality rate/(μg/m³) for mortality.

We have also assessed the impacts of $PM_{2.5}$ as an alternative to PM_{10}. The functions used are based on the same studies as the PM_{10} functions, though they have different slope factors. In fact, many of the functions presented are from studies where the pollution increment measures were originally $PM_{2.5}$ (and we have converted the functions into PM_{10} for the core assessment). The functions used are shown in Table 8.65.

The damage estimates of morbidity effects from $PM_{2.5}$ concentrations are shown in Table 8.66, whereas the cases are shown in the Appendix in Table A.8.36.

Acute and chronic mortality physical impacts damages caused by $PM_{2.5}$ are reported in the Appendix in Tables A.8.37 and A.8.38, whereas the respective monetary damages are shown in Tables 8.67 and 8.68. The values presented are the total damages calculated with the function, and so also include acute mortality damages.

Table 8.65 Quantification of human health impacts for $PM_{2.5}$

Receptor	Impact category	Reference	f_{er}	Uncertainty rating
Asthmatics (3.5% of population)				
Adults	Bronchodilator usage	Dusseldorp et al., 1995	0.272	B
	Cough	Dusseldorp et al., 1995	0.280	A
	Lower respiratory symptoms	Dusseldorp et al., 1995	0.101	A
Children	Bronchodilator usage	Roemer et al., 1993	0.129	B
	Cough	Pope and Dockery, 1992	0.223	A
	Lower respiratory symptoms	Roemer et al., 1993	0.172	A
Elderly 65+				
	Congestive heart failure	Schwartz and Morris, 1995	3.09E-5	B
Children				
	Chronic bronchitis	Dockery et al., 1989	2.69E-3	B
	Chronic cough	Dockery et al., 1989	3.46E-3	B
Adults (80% of population)				
	Restricted activity days (RAD)	Ostro, 1987	0.042	B
	Chronic bronchitis	Abbey et al., 1995	7.8E-5	A
Entire population				
	Respiratory hospital admissions	Dab et al., 1996	3.46E-6	A
	Cerebrovascular hosp. adm.	Wordley et al., 1997	8.42E-6	B
	Acute mortality (AM)	Verhoeff et al., 1996	0.068%	B

Table 8.66 Estimated monetary value for morbidity from $PM_{2.5}$ in the UK in 1996

$PM_{2.5}$ morbidity		Human morbidity (million Euro)			
		Background $PM_{2.5}$			
	Cost per unit (Euro)	0 $\mu g/m^3$	3 $\mu g/m^3$	6 $\mu g/m^3$	9 $\mu g/m^3$
Congest. heart failure in adults > 65 years	7870	29.33	22.89	16.44	9.99
Chronic bronchitis in adults	240 000	11384.59	8895.61	6406.63	3917.64
Restricted activity days in adults	75	1915.68	1496.86	1078.04	659.22
Bronchodilator usage in adult asthmatics	7.5	62.03	48.47	34.91	21.35
Cough days in adult asthmatics	7.5	63.86	49.90	35.93	21.97
Lower resp. symp. days in adult asthmatics	7.5	23.03	18.00	12.96	7.93
Bronchodilator usage in child asthmatics	7.5	14.89	11.62	8.35	5.08
Cough days in child asthmatics	7.5	25.74	20.09	14.43	8.78
Lower resp. symp. days in child asthmatics	7.5	19.85	15.49	11.13	6.77
Case of chronic bronchitis in children	225	93.14	72.69	52.23	31.78
Chronic cough episode in children	225	119.81	93.50	67.19	40.88
Cerebrovascular hosp. adm. in total pop.	7870	50.49	39.44	28.40	17.35
Resp. hospital admissions in total pop.	7870	20.75	16.21	11.67	7.13

Table 8.67 Estimated monetary value for acute mortality from $PM_{2.5}$ in the UK in 1996

$PM_{2.5}$ acute mortality	Human mortality (million Euro)			
	Background $PM_{2.5}$			
	0 μg/m³	3 μg/m³	6 μg/m³	9 μg/m³
Low 147 000 Euro (3%)*0.75	645	504	363	222
High 310 500 Euro (11%)*0.75	1364	1065	767	468

Table 8.68 Estimated monetary value for chronic mortality from $PM_{2.5}$ in the UK in 1996

PM2.5 chronic mortality*	Human mortality (million Euro)			
	Background $PM_{2.5}$			
	0 μg/m³	3 μg/m³	6 μg/m³	9 μg/m³
Low (11%)	12 700	9915	7131	4346
High (3%)	30 480	23 797	17 114	10 431

Note: *Using values of 10 and 24 million Euro per one-year reduction of 6μg/m³ $PM_{2.5}$ per 100 000 people.

The comparison of the two metrics is shown in Tables 8.69 and 8.70. In all cases, the values presented for chronic mortality are the total damages calculated with the function, and so also include acute mortality damages.

The results for acute mortality and PM_{10}/$PM_{2.5}$ are very similar, with the $PM_{2.5}$ values being slightly larger. The difference comes in the use of the original $PM_{2.5}$ function to quantify chronic mortality from $PM_{2.5}$ levels. The chronic mortality values for $PM_{2.5}$ are very much greater than the damages from PM_{10} (using the central estimate). This is because the central estimate converts to PM_{10} from the original Pope sulphate functions. If instead the values for PM_{10} are derived from the original Pope $PM_{2.5}$ function, then we get very similar results.

8.5 CONCLUSIONS

The main analysis (Section 8.3) includes the estimation of damages on human health, crops and material in Germany, Italy and the Netherlands

Table 8.69 Estimated $PM_{10}/PM_{2.5}$ mortality impacts in the UK in 1996

Mortality	Human mortality (cases)	
	0 μg/m³	Background
Acute mortality PM_{10}	5288	2771 (10 μg/m³)
Acute mortality $PM_{2.5}$	5855	3293 (6 μg/m³)
	Human mortality (YOLL)	
Chronic mortality PM_{10} (1)	261 861	137 221 (10 μg/m³)
Chronic mortality PM_{10} (2)	549 909	288 165 (10 μg/m³)
Chronic mortality $PM_{2.5}$ (3)	596 890	335 147 (6 μg/m³)

Notes:
(1) Using values of 10 and 24 million Euro per one-year reduction of 21μg/m³ PM_{10} per 100 000 people.
(2) Using values of 10 and 24 million Euro per one-year reduction of 10μg/m³ PM_{10} per 100 000 people.
(3) Using values of 10 and 24 million Euro per one-year reduction of 6μg/m³ $PM_{2.5}$ per 100 000 people.

Table 8.70 Estimated monetary value for $PM_{10}/PM_{2.5}$ mortality in the UK in 1996

Mortality	Human mortality (million Euro)				
	No background		Background		Background
Discount rate	3%	11%	3%	11%	level
Acute mortality PM_{10}	583	1232	306	645	(10 μg/m³)
Acute mortality $PM_{2.5}$	645	1364	363	767	(6 μg/m³)
Chronic mortality PM_{10} (1)	12 594	3929	6600	2059	(10 μg/m³)
Chronic mortality PM_{10} (2)	28 080	11 700	14 715	6131	(10 μg/m³)
Chronic mortality $PM_{2.5}$ (3)	30 480	12 700	17 114	7131	(6 μg/m³)

Notes:
(1) Using values of 10 and 24 million Euro per one-year reduction of 21μg/m³ PM_{10} per 100 000 people.
(2) Using values of 10 and 24 million Euro per one-year reduction of 10μg/m³ PM_{10} per 100 000 people.
(3) Using values of 10 and 24 million Euro per one-year reduction of 6μg/m³ $PM_{2.5}$ per 100 000 people.

for 1994 and in the UK for 1996 by applying the recommended exposure–response functions and monetary values reported in Chapters 4 and 5 respectively. Although large differences exist in the databases and therefore in the pollutants which could be taken into account for the four country studies, the effects of PM_{10} on human health, which represent the largest share of all damage costs in all countries, as well as the human health impacts of SO_2, could be estimated for each country. Concerning these effects, estimated damage costs were 40.2 billion Euro p.a. in Germany, 44.5 billion Euro p.a. in Italy, 9.9 billion Euro p.a. in the Netherlands in 1994 and 14.0 billion Euro p.a. in the UK in 1996. The largest damages referenced to GDP were for Italy (4.4 per cent) followed by the Netherlands (3.6 per cent), Germany (2.5 per cent) and the UK (1.5 per cent). The same ranking is observed for the costs per inhabitant, which are about 780 Euro per person and year in Italy, 640 Euro per person and year in the Netherlands, 500 Euro per person and year in Germany and 240 Euro per person and year in the UK. For the countries in which also damages caused by O_3 on human health and by O_3 and SO_2 on crops and material were assessed, it could be seen that these damage costs are about one order of magnitude smaller than the human health damage costs effected by PM_{10}.

The analysis for the UK (Section 8.4) shows that by 1995 damage costs had fallen to 80 per cent of those estimated for 1990. A larger effect is expected for the year on year changes in Germany, because particle emissions were reduced by a factor of 2.5 from 1990 to 1994 due to the modernisation of power plants and production processes in the eastern parts of Germany.

Sensitivity analysis on further pollutants like CO, benzene/butadiene and heavy metals suggests that these are much less significant than the damage costs caused by O_3, SO_2 and particles. Furthermore, impacts of O_3 on rubber and paint and soiling effects due to PM_{10} were investigated. These effects, which were assessed at about 120 and 335 million Euro p.a., are a significant addition to the material damages estimated in the core analysis. The calculation of alternative exposure–response in a sensitivity analysis shows that the aggregated effects calculated for morbidity impacts are very small and can therefore be neglected while the impacts on acute mortality concerning NO_2 are as high as the acute mortality effects caused by PM_{10}. The acute mortality effects of CO are slightly lower.

In the present study damage costs for impacts of air pollution on crop yields, building materials, and human health were estimated and compared for the four analysed countries to show the different environmental situations. The analysis of the year on year changes in damage costs demonstrates that the methodology can also be used to report the environmental development in one country concerning damages caused by air pollution. Thereby the impact pathway analysis can be applied to verify the effectiveness of national and international political measures in this area.

The damage estimation includes uncertainties, concerning the interpolation, the background levels and especially concerning the exposure–response functions and monetary valuation. Analysis of all pollutants could not be included in the main analysis, because the environmental data needed were not available in all countries. For most of these pollutants, a sensitivity analysis has shown that effects are negligible compared to the damages in the main analysis. It is stressed that not all damage costs have been included. These especially concern impacts on very complex processes in some environmental areas, for example, in natural ecosystems and forests for which reliable exposure–response relations are not available. Instead of a monetary valuation the regions in which critical levels were exceeded were specified in some country studies. Large areas of critical level exceedances show that impacts on natural vegetation and forest ecosystems may be important. More research is needed to achieve progress in exposure–response functions and monetary valuation for these impacts.

NOTES

1. AOT60 means Accumulated Ozone Concentration above a threshold of 60 ppbVh.
2. AOT40 means Accumulated Ozone Concentration above a threshold of 40 ppbVh.
3. Its website is http://www.etcaq.rivm.nl/airbase.
4. In AOT40c the letter ‘c’ stands for crop.

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APPENDIX 8.1: PHYSICAL IMPACTS

Germany

Health – morbidity

Table A.8.1 Estimated effects on morbidity caused by SO_2 in Germany in 1994

Function	Impact (1000*unit p.a.)		
	Background		
	0 ppbV	1 ppbV	2 ppbV
Resp. hosp. adm. in total population (case)	2.3	1.9	1.5

Table A.8.2 Estimated effects on morbidity from PM_{10} in Germany in 1994

Function	Impact (1000*unit p.a.)			
	Background			
	0 μg/m3	5 μg/m^3	10 μg/m^3	15 μg/m^3
Congestive heart failure in adults aged over 65 years (case)	6.2	5.2	4.2	3.1
Chronic bronchitis in adults (case)	100	85	68	51
Restricted activity days in adults (day)	52 000	43 000	35 000	26 000
Bronchodilator usage in asthm. (ad.) (case)	8400	7000	5600	4200
Cough in asthm. (ad.) (day)	8600	7200	5800	4400
Lower resp. symptoms in asthm. (ad.) (day)	3100	2600	2100	1600
Bronchodilator usage in asthm. (ch.) (case)	1700	1400	1100	850
Cough in asthm. (ch.) (day)	2900	2400	1900	1500

Table A.8.2 (continued)

Function	Impact (1000*unit p.a.)			
	Background			
	0 μg/m³	5 μg/m³	10 μg/m³	15 μg/m³
Lower resp. symptoms in asthm. (ch.) (day)	2200	1900	1500	1100
Case of chronic bronchitis in children (case)	1000	830	670	500
Chronic cough in children (episode)	1300	1100	860	650
Cerebrov. hosp. admissions in total pop. (case)	13	11	8.7	6.6
Resp. hosp. admissions in total pop. (case)	5.4	4.5	3.6	2.7

Table A.8.3 *Estimated effects on morbidity caused by O_3 in Germany in 1994*

Function	Impact (1000*unit p.a.)			
	Background			
	0 ppbV	16 ppbV	20 ppbV	24 ppbV
Asthma attacks in all asthmatics (case)	8 900	4 500	3 400	2 300
Minor restr. activity days in adults (day)	41 000	21 000	15 000	10 000
Symptom days in total population (day)	170 000	86 000	65 000	44 000
Resp. hosp. admissions in total population (case)	18	9.3	7	4.7

Note: Damages are estimated from six-hour annual mean concentration in 1994.

Table A.8.4 Rough estimated damages caused by cadmium concentration in Germany in 1994

Function	Impacts (cases p.a.)	Damages (million Euro p.a.)[a]
Cancer in total population, (US EPA cadmium unit risk)	0.57	1.8

Note: [a] Calculated as fatal cancer with the value of statistical life of 3.1 million Euro.

Health – mortality

Table A.8.5 Estimated acute effects on mortality caused by anthropogenic SO_2 concentration in Germany in 1994

Function	Acute mortality, YOLL (years lost p.a.)		
	Background		
	0 ppbV	1 ppbV	2 ppbV
Acute mortality, YOLL in total population	6200	5000	3900

Table A.8.6 Estimated acute effects on mortality caused by anthropogenic O_3 concentration in Germany in 1994

Function	Acute mortality (years lost p.a.)			
	Background			
	0 ppbV	16 ppbV	20 ppbV	24 ppbV
Acute mortality, YOLL in total population	23000	11600	8700	5900

Table A.8.7 Estimated acute and chronic effects on mortality caused by fine particles in Germany in 1994

Function	Chronic mortality (years lost per annum)			
	Background			
	0 μg/m³	5 μg/m³	10 μg/m³	15 μg/m³
Mortality, YOLL (99% chronic) in total population	580 000	490 000	390 000	290 000

Note: It was assumed that 85 per cent of the total suspended matter are smaller than 10 μm (VDI, 1990).

Crops

Table A.8.8 Estimated impacts resulting from effects of SO_2 and O_3 on crops in Germany in 1994

Function	Yield Loss (1000t per annum)			
	O_3	SO_2		
	(AOT40 crops)	Background		
		0 ppbV	1 ppbV	2 ppbV
Barley	2600	−220	−120	−45
Oats	180	−31	−20	−9.8
Potato	5800	−450	−280	−130
Rye	230	−49	−32	−19
Sugar beet	N.A.	−160	−93	−40
Sunflower seed	6.2e-1	N.A.	N.A.	N.A.
Tobacco	5.2	N.A.	N.A.	N.A.
Wheat	3500	−260	−150	−52

Materials

Table A.8.9 Estimated surface areas which had to be repaired in Germany in 1994, resulting from incremental effects of SO_2 and O_3 on building material

Function	Maintenance surface (m^2/a)			
	Natural background for SO_2/O_3			
	0/0 ppbV	0/16 ppbV	1/20 ppbV	2/24 ppbV
SO_2				
Limestone	6 700	6 700	5 300	3 900
Natural stone	7 000	7 000	5 500	4 100
Paint	10 390 000	10 390 000	8 240 000	6 080 000
Rendering	660 000	660 000	520 000	390 000
Sandstone	10 000	10 000	8 100	5 900
SO_2 and O_3				
Galvanised steel	1 330 000	660 000	390 000	190 000
Zinc	180 000	87 000	53 000	26 000

Italy

Health – mortality and morbidity

Table A.8.10 Estimated impacts resulting from effects of SO_2 on human health in Italy in 1994

Function		Cases/year (YOLL/year for mortality)		
		SO_2 – Background (ppbV)		
Receptor	Impact category	0	1	2
Entire population				
	Respiratory hospital admissions	1708	1407	1124
	Acute mortality	4295	3538	2827

Table A.8.11 Estimated impacts resulting from mortality effects of PM_{10} on human health in Italy in 1994

Function		Cases/year (YOLL/year for mortality)				
		PM_{10} – Background (μg/m^3)				
Receptor	Impact category	0	5	10	15	20
Asthmatics (3.5% of population)						
Adults	Bronchodilator usage	11 354 743	10 067 437	8 780 132	7 492 826	6 205 520
	Cough	11 703 048	10 376 254	9 049 461	7 722 667	6 395 874
	Lower respiratory symptoms (wheeze)	4 249 321	3 767 569	3 285 816	2 804 064	2 322 311
Children	Bronchodilator usage	1 358 389	1 204 387	1 050 384	896 381	742 378
	Cough	2 316 228	2 053 634	1 791 039	1 528 445	1 265 850
	Lower respiratory symptoms (wheeze)	1 793 771	1 590 408	1 387 045	1 183 683	980 320
Elderly 65+ (14% of population)						
	Congestive heart failure	6 444	5 713	4 983	4 252	3 522
Children (20% of population)						
	Chronic bronchitis	801 101	710 279	619 457	528 635	437 813
	Chronic cough	1 029 988	913 216	796 445	679 674	562 902
Adults (80% of population)						
	Restricted activity days	49 757 857	44 116 728	38 475 599	32 834 470	27 193 341
	Chronic bronchitis	97 525	86 469	75 412	64 356	53 299
Entire population						
	Respiratory hospital admissions	5 150	4 566	3 982	3 398	2 815
	Cerebrovascular hospital admissions	12 539	11 117	9 696	8 274	6 853
	Acute mortality	7 090	6 287	5 483	4 679	3 875
	Chronic mortality	556 814	493 687	430 560	367 433	304 306

Table A.8.12 Estimated impacts resulting from mortality effects of O_3 on human health in Italy in 1994

Function		Cases/year (YOLL/year for mortality)			
		O_3 – Background (ppbV)			
Receptor	Impact category	0	16	20*	24*
Asthmatics (3.5% of population)					
All asthma attacks		2 814 802	157 206	0	0
Adults (80% of population)					
	Minor restricted activity days	14 637 345	817 493	0	0
Entire population					
	Respiratory hospital admissions	6 636	371	0	0
	Symptom days	61 863 780	3 455 079	0	0
	Acute mortality	7 881	440	0	0

Note: * No impacts are reported, because the average O_3 concentration is below the background value.

Crops

Table A.8.13 Estimated impacts resulting from effects of SO_2 on crops in Italy in 1994

Function Baker et al. (1986) modified	Yield loss (kt)		
	SO_2 – Background		
	0 ppbV	1 ppbV	2ppbV
Barley	39.8	28.2	18.5
Oats	0.4	0.3	0.2
Potato	42.3	27.7	15.5
Rye	8.0	5.7	3.7
Sugar beet	238.0	160.4	95.7
Wheat	163.6	102.8	52.1

The Netherlands

Health – mortality and morbidity

Table A.8.14 Morbidity impacts due to SO_2, PM_{10}, O_3 and CO air pollution in the Netherlands in 1994 exceeding different natural background concentrations

Receptor	Health impact	Impact (cases per year)			
		Background			
		zero	low	mid	high
		SO_2			
		0 ppbV	0 ppbV	1 ppbV	2 ppbV
Entire population	Respiratory hospital admission	290	290	210	130
		PM_{10}			
		0 $\mu g/m^3$	5 $\mu g/m^3$	10 $\mu g/m^3$	15 $\mu g/m^3$
Asthmatics					
Adults	Bronchodilator usage	1 300 000	1 100 000	960 000	790 000
	Cough	1 300 000	1 200 000	990 000	810 000
	Lower respiratory symptoms	490 000	420 000	360 000	300 000
Children	Bronchodilator usage	140 000	120 000	100 000	85 000
	Cough	240 000	210 000	180 000	150 000
	Lower respiratory symptoms	190 000	160 000	140 000	110 000
Elderly 65+	Congestive heart failure	1400	1200	1000	830
Children 15-	Bronchitis	170 000	150 000	130 000	100 000
	Cough	220 000	190 000	160 000	130 000
Adults 15+	Chronic bronchitis	23 000	20 000	17 000	14 000
	Net restricted activity days	12 000 000	10 000 000	8 500 000	7 000 000
Entire population	Respiratory hospital admissions	1 200	1 000	880	720

Table A.8.14 (continued)

Receptor	Health impact	Impact (cases per year)			
		Background			
		zero	low	mid	high
		PM_{10}			
		0 μg/m³	5 μg/m³	10 μg/m³	15 μg/m³
	Cerebrovascular hospital admissions	2 900	2 500	2 100	1 800
		O3			
		0 ppbV	16 ppbV	20 ppbV	24 ppbV
Asthmatics					
	All asthma attacks	550 000	190 000	100 000	16 000
Entire population	Respiratory hospital admissions	2 700	940	510	76
	Symptom days	25 000 000	8 800 000	4 700 000	710 000
Adults 15+	Minor restricted activity days	5 900 000	2 000 000	1 000 000	32 000
		CO			
		0 ppbV	0.15 ppbV		
Elderly 65+	Congestive heart failure	530	330		

Table A.8.15 Cancer, inhalation and IQ (Intelligence Quotient) health impacts due to heavy metal air pollution in the Netherlands in 1994

Receptor	Health impact	Pollutant	Impact (cases per year)
			Background level 0 ng/m^3
Entire population	Cancer	As	1.5
	Cancer	Cd	0.22
	Cancer	Cr	nq
	Cancer	Ni	nq
	Cancer	Hg	nq
	Cancer	Pb	nq
	Inhalation effects	Hg	nq
Children 2–	IQ points in children	Pb	17 000

Note: nq: Not quantified.

Table A.8.16 Mortality impacts due to PM_{10}, SO_2 and O_3 air pollution in the Netherlands in 1994

Receptor	Health impact	Impact (cases/y)			
		Background			
		zero	low	mid	high
		SO_2			
		0 ppbV	0 ppbV	1 ppbV	2 ppbV
Entire population	Acute mortality	840	840	610	380
		PM_{10}			
		0 μg/m^3	5 μg/m^3	10 μg/m^3	15 μg/m^3
Entire population	Acute mortality	1 900	1 600	1 400	1 100
	Chronic mortality *	120 000	110 000	92 000	75 000

Table A.8.16 (continued)

Receptor	Health impact	Impact (cases/y)			
		Background			
		zero	low	mid	high
		O_3			
		0 ppbV	16 ppbV	20 ppbV	24 ppbV
Entire population	Acute mortality	3 600	1 300	690	100
Subtotal	Acute mortality	6 300	3 700	2 700	1 600
	Chronic mortality*	120 000	110 000	92 000	75 000

Note: * It was assumed that chronic mortality impacts have an average delay of ten years.

Crops

Table A.8.17 Crop yield loss in kt/y due to O_3 and SO_2 air pollution in the Netherlands in 1994 exceeding different natural background concentrations

Receptor	Yield loss (kt/y)			
	AOT40c	SO_2		
		Background level		
		0 ppbV	1 ppbV	2 ppbV
Barley	0	−3.5	−5.5	−7.7
Oats	2.3	−0.3	−0.59	−0.87
Potato	1051	−98	−158	−227
Rye	3.4	−0.5	−0.8	−1.2
Sugar beet	0	−103	−161	−228
Wheat	145	−14	−22	−32
Rice	0	0	0	0
Sunflower seed	–	N.A.	N.A.	N.A.
Tobacco	0	0	0	0
Peas and beans	–	0.99	0.64	0.30

Notes:
N.A. = Not analysed.
– = No exposure–response function available.

Table A.8.18 Crop yield loss in percentage of total production due to O_3 and SO_2 air pollution in the Netherlands in 1994 exceeding different natural background concentrations

Receptor	Yield loss (% total production)			
	AOT40c	SO_2		
		Background level		
		0 ppbV	1 ppbV	2 ppbV
Barley	0	−1.4	−2.2	−3.1
Oats	7.5	−1.1	−1.9	−2.8
Potato	12.4	−1.1	−1.9	−2.7
Rye	8.1	−1.1	−1.9	−2.8
Sugar beet	0	−1.4	−2.2	−3.1
Wheat	14.1	−1.4	−2.2	−3.1
Rice	0	0	0	0
Sunflower seed	–	N.A.	N.A.	N.A.
Tobacco	0	0	0	0
Peas and beans	–	2.1	1.4	0.6

Notes:
N.A. = Not analysed.
– = No exposure–response function available.

Materials

Table A.8.19 Impacts on materials due to air pollutants in the Netherlands in 1994 exceeding different natural background concentrations

Receptor	Impact (1000 m^2/y)			
	Background level			
	SO_2	0 ppbV	1 ppbV	2 ppbV
	O_3	16 ppbV	20 ppbV	24 ppbV
Zinc		4.8	−0.29	−2.6
Galvanised steel		239	−15	−128
Limestone		1.1	0.8	0.5
Sandstone		0.73	0.53	0.32
Natural stone		0.77	0.42	0.071

Table A.8.19 (continued)

Receptor	Impact (1000 m^2/y)		
	Background level		
SO_2	0 ppbV	1 ppbV	2 ppbV
O_3	16 ppbV	20 ppbV	24 ppbV
Mortar	–	–	–
Rendering	71	51	31
Paint	1095	791	487

The United Kingdom

Health – mortality and morbidity

Table A.8.20 Estimated morbidity impacts resulting from SO_2 in the UK in 1996

SO_2 morbidity	Human morbidity (cases)		
	Background SO_2		
Health (morbidity)	0 ppbV	1 ppbV	2 ppbV
Resp. hosp. admissions in total population *Ponce de Leon, 1996*	1255	952	650

Table A.8.21 Estimated acute mortality impacts resulting from SO_2 in the UK in 1996

SO_2 acute mortality	Human mortality (cases)		
	Background SO_2		
Health (mortality)	0 ppbV	1 ppbV	2 ppbV
Acute mortality *Anderson et al., 1996; Touloumi et al., 1996*	5003	3798	2593

Table A.8.22 Estimated morbidity impacts resulting from PM_{10} in the UK in 1996

PM_{10} morbidity	Human morbidity (cases)			
	Background PM_{10}			
	0 μg/m³	5 μg/m³	10 μg/m³	15 μg/m³
Congest. heart failure in adults > 65 years *Schwartz and Morris, 1995*	3433	2614	1795	976
Chronic bronchitis in adults *Abbey et al., 1995*	45 753	34 873	23 993	13 113
Restricted activity days in adults *Ostro, 1987*	23 343 219	17 792 202	12 241 185	6 690 168
NET restricted activity days in adults* *Ostro, 1987*	23 029 603	17 553 214	12 076 880	6 600 491
Bronchodilator usage in adult asthmatics *Dusseldorp et al., 1995*	7 609 889	5 800 258	3 990 626	2 180 995
Cough days in adult asthmatics *Dusseldorp et al., 1995*	7 843 322	5 978 180	4 113 038	2 247 896
Lower resp. sympt. days in adult asthmatics *Dusseldorp et al., 1995*	2 847 873	2 170 649	1 493 425	816 200
Bronchodilator usage in child asthmatics *Roemer et al., 1993*	1 843 799	1 403 576	963 353	523 130
Cough days in child asthmatics *Pope and Dockery, 1992*	3 143 913	2 393 277	1 642 640	892 003
Lower resp. symp. days in child asthmatics *Roemer et al., 1993*	2 434 760	1 853 440	1 272 120	690 800
Cases of chronic bronchitis in children *Dockery et al., 1989*	380 579	289 712	198 846	107 979
Chronic cough episodes in children *Dockery et al., 1989*	489 316	372 487	255 659	138 831
Cerebrovascular hosp. adm. in total pop. *Wordley et al., 1997*	5897	4494	3090	1687

Table A.8.22 (continued)

PM_{10} morbidity	Human morbidity (cases)			
	Background PM_{10}			
	0 $\mu g/m^3$	5 $\mu g/m^3$	10 $\mu g/m^3$	15 $\mu g/m^3$
Resp. hospital admissions in total pop. *Dab et al., 1996*	2 422	1 846	1 269	693

Note: * Restricted activity days (RAD) are also presented as net cases in the table. This assumes that all days in hospital for respiratory admissions (RHA), congestive heart failure (CHF) and cerebrovascular conditions (CVA) (assuming that the average stay for each is 10, 7 and 45 days respectively) are subtracted from the total restricted activity days (RAD).

Table A.8.23 *Estimated acute mortality impacts resulting from PM_{10} in the UK in 1996*

PM_{10} acute mortality	Human mortality (cases)			
	Background PM_{10}			
	0 $\mu g/m^3$	5 $\mu g/m^3$	10 $\mu g/m^3$	15 $\mu g/m^3$
Acute mortality *Spix and Wichmann, 1996*	5288	4030	2771	1513

Table A.8.24 *Estimated chronic mortality impacts resulting from PM_{10} in the UK in 1996*

PM_{10} chronic mortality	Human mortality (YOLL)			
	Background PM_{10}			
	0 $\mu g/m^3$	5 $\mu g/m^3$	10 $\mu g/m^3$	15 $\mu g/m^3$
Chronic mortality *Pope et al., 1995*	261 861	199 541	137 221	74 902

Table A.8.25 Estimated morbidity impacts resulting from O_3 in the UK in 1996

O_3 morbidity	Human morbidity (cases)			
	Background O_3			
	0 ppbV	16 ppbV	20 ppbV	24 ppbV
Minor restricted activity days in adults *Ostro and Rothschild, 1989*	25 958 097	12 088 548	8 621 160	5 153 773
NET minor rest. activity days in adults* *Ostro and Rothschild, 1989*	25 817 935	12 023 309	8 574 651	5 153 773
Asthma attacks in all asthmatics *Whittemore and Korn, 1980*	5 005 768	2 329 969	1 661 019	992 069
Resp. hosp. adm. in total population *Ponce de Leon et al., 1996*	11 802	5493	3916	2339
Symptom days in total population *Krupnick et al., 1990*	110 016 885	51 208 101	36 505 905	21 803 708

Table A.8.26 Estimated acute mortality impacts resulting from O_3 in the UK in 1996

O_3 acute mortality	Human mortality (cases)			
	Background O_3			
	0 ppbV	16 ppbV	20 ppbV	24 ppbV
Acute mortality *Sunyer et al., 1996*	22 227	10 346	7 375	4 405

Materials

Table A.8.27 Estimated impacts from effects of SO_2 and acidity on material in the UK in 1996

Material	Material loss (m^2)	
	Background	
	SO_2	Acidity
	1 ppbV	pH 5
Natural stone	33 556	6 369
Mortar	494 097	79 835
Paint (carbonate)	34 970 452	3 257 316
Galvanised steel	22 188 361	1 967 014

Sensitivity Analysis

Table A.8.28 Estimated morbidity impacts from CO in the UK in 1996 (sensitivity)

CO morbidity	Human morbidity (cases)	
	Background CO	
	0 ppmV	0.15 ppmV
Congest. heart failure in adults > 65 years *Schwartz and Morris, 1995*	1588	733

Table A.8.29 Estimated cancer risk from benzene and 1.3-butadiene in the UK in 1996 (sensitivity)

Cancer risk	Pilkington et al., 1997	Human morbidity (cases)
	URF	Background
	$\mu g/m^3$ per year	0 $\mu g/m^3$
Benzene	1.14E−7	14
1.3-butadiene	4.29E−6	70

Table A.8.30 Estimated morbidity impacts resulting from PM_{10} in the UK in 1996 (sensitivity)

PM_{10} morbidity	Human morbidity (cases)			
	Background PM_{10}			
	0 μg/m^3	5 μg/m^3	10 μg/m^3	15 μg/m^3
Ischaemic heart disease *Schwartz and Morris, 1995*	3 247	2 473	1 698	923
ERV for COPD *Sunyer et al., 1993*	8 424	6 419	4 414	2 410
ERV for asthma *Schwartz, 1993; Bates, 1990*	7 547	5 751	3 955	2 159
ERV for croup in pre-school children *Schwartz et al., 1991*	34 048	25 945	17 842	9 739

Table A.8.31 Estimated morbidity impacts resulting from O_3 in the UK in 1996 (sensitivity)

O_3 morbidity	Human morbidity (cases)			
	Background O_3			
	0 ppbV	16 ppbV	20 ppbV	24 ppbV
ERV for asthma *Cody et al., 1992; Bates et al., 1990*	44 007	20 483	36 656	32 980

Table A.8.32 Estimated morbidity impacts from NO_2 in the UK in 1996 (sensitivity)

NO_2 morbidity	Human morbidity (cases)	
	Background NO_2	
	0 ppmV	2 ppmV
Respiratory hospital admissions *Ponce de Leon et al., 1996*	2073	1775

Table A.8.33 Estimated acute mortality impacts from NO_2 in the UK in 1996 (sensitivity)

NO_2 acute mortality	Human mortality (cases)	
	Background NO_2	
	0 ppmV	2 ppmV
Acute mortality *Sunyer et al., 1996;* *Anderson et al., 1996*	5689	5559

Table A.8.34 Estimated morbidity impacts from CO in the UK in 1996 (sensitivity)

CO morbidity	Human morbidity (cases)	
	Background CO	
	0 ppmV	0.15 ppmV
Ischaemic heart failure in adults > 65 years *Schwartz and Morris, 1995*	1193	551

Table A.8.35 Estimated acute mortality impacts resulting from CO in the UK in 1996 (sensitivity)

CO acute mortality	Human mortality (cases)	
	Background CO	
	0 ppmV	0.15 ppmV
Acute mortality *Touloumi et al., 1994*	3116	1473

Table A.8.36 Estimated morbidity impacts resulting from $PM_{2.5}$ in the UK in 1996

$PM_{2.5}$ morbidity	Human morbidity (cases)			
	Background $PM_{2.5}$			
	0 μg/m³	3 μg/m³	6 μg/m³	9 μg/m³
Congest. heart failure in adults > 65 years *Schwartz and Morris, 1995*	3727	2908	2089	1270
Chronic bronchitis in adults *Abbey et al., 1995*	47 436	37 065	26 694	16 324
Restricted activity days in adults *Ostro, 1987*	25 542 352	19 958 095	14 373 838	8 789 581
Bronchodilator usage in adult asthmatics *Dusseldorp et al., 1995*	8 270 857	6 462 621	4 654 386	2 846 150
Cough days in adult asthmatics *Dusseldorp et al., 1995*	8 514 117	6 652 698	4 791 279	2 929 860
Lower resp. sympt. days in adult asthmatics *Dusseldorp et al., 1995*	3 071 164	2 399 723	1 728 283	1 056 843
Bronchodilator usage in child asthmatics *Roemer et al., 1993*	1 985 219	1 549 254	1 113 289	677 325
Cough days in child asthmatics *Pope and Dockery, 1992*	3 431 813	2 678 168	1 924 524	1 170 879
Lower resp. symp. days in child asthmatics *Roemer et al., 1993*	2 646 959	2 065 672	1 484 386	903 100
Cases of chronic bronchitis in children *Dockery et al., 1989*	413 972	323 062	232 151	141 241
Chronic cough episodes in children *Dockery et al., 1989*	532 470	415 536	298 603	181 670
Cerebrovascular hosp. adm. in total pop. *Wordley et al., 1997*	6416	5012	3608	2204
Resp. hospital admissions in total pop. *Dab et al., 1996*	2636	2060	1483	906

Table A.8.37 Estimated acute mortality impacts resulting from $PM_{2.5}$ in the UK in 1996

$PM_{2.5}$ acute mortality	Human mortality (cases)			
	Background $PM_{2.5}$			
	0 μg/m^3	3 μg/m^3	6 μg/m^3	9 μg/m^3
Acute mortality *Spix and Wichmann, 1996*	5855	4574	3293	2011

Table A.8.38 Chronic mortality impacts resulting from $PM_{2.5}$ in the UK in 1996

$PM_{2.5}$ chronic mortality	Human mortality (cases)			
	Background $PM_{2.5}$			
	0 μg/m^3	3 μg/m^3	6 μg/m^3	9 μg/m^3
Chronic mortality *Pope et al., 1995*	596 890	466 018	335 147	204 275

9. Attribution of air damages to countries and economic sectors of origin

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9.1 INTRODUCTION

This chapter deals with the application of the impact pathway approach, which was developed under the ExternE project (EC, 1995) for the calculation of damages caused by air pollution in the EU and the attribution of damages by economic activity. Here, calculations are made using modelled emissions. In the first part of the present chapter the impact and damage cost assessment used is reported in more detail. Following that (in Sections 9.3 to 9.5) the emission scenarios and the corresponding results are presented. Section 9.6 includes a further analysis of the results. Finally, in Section 9.7 the results obtained with modelled data are compared with those obtained with measured concentration data (as presented in Chapter 8).

9.2 METHODOLOGY OF THE IMPACT AND DAMAGE COST ASSESSMENT

This part of the study concentrates on the effects on crop yields, building materials and human health of emissions of sulphur dioxide (SO_2), nitrogen oxides (NO_X), ammonia (NH_3), non-methane volatile organic compounds (NMVOC) and related secondary pollutants like ozone (O_3) and particles. An estimation of the impacts of directly emitted particles smaller than 10 micrometres (PM_{10}) was also carried out. Note that pollutant estimates are not calculated using the same sources as in the previous chapter, but from CORINAIR. The exposure–response functions and monetary values discussed in Chapters 4 and 5 were used for the impact assessment. The stock at risk data of crop cultivation and population distribution used

Table 9.1 Environmental effects included in the analysis

Impact category	Pollutant	Effects
Human health – mortality	Particles, SO_2, ozone	Reduction of life expectancy due to short- and long-term exposure
Human health – morbidity	Particles, ozone	Respiratory hospital admissions Restricted activity days
	Particles	Cardiovascular hospital admissions Cases of chronic bronchitis Cases of chronic cough in children Cough in asthmatics Lower respiratory symptoms
	O_3	Asthma attacks
Materials	SO_2, acid deposition	Ageing of galvanised steel, limestone, mortar, sandstone, paint, rendering and zinc for utilitarian buildings
Crops	SO_2	Yield changes for wheat, barley, rye, oats, potato, sugar beet
	Ozone	Yield changes for wheat, barley, rye, oats, potato, rice, tobacco, sunflower seed
	Acid deposition	Increased need for liming
	N, S	Fertilising effects

for the study are based on European statistical data for 1993/94 (Eurostat, 1995), while the material inventory was extrapolated from studies carried out for different European cities (ECOTEC, 1986; Hoos et al., 1987; Kucera et al., 1993; Tolstoy et al., 1990 (see also EC, 1997b; Mayerhofer et al., 1997). Table 9.1 shows the physical effects included in the current analysis.

9.3 EMISSION SCENARIOS

The present analysis of damage attribution includes the estimation of impacts and damage costs caused by economic activities and their geographical distribution. The different emission scenarios used are described below.

Table 9.2 CORINAIR 1990 main economic sectors

SNAP unit	Economic sector
1	Public power, cogeneration and district heating plants
2	Commercial, institutional and residential combustion plants
3	Industrial combustion
4	Production processes
5	Extraction and distribution of fossil fuels
6	Solvent use
7	Road transport
8	Other mobile sources and machinery
9	Waste treatment and disposal
10	Agriculture
11	Nature

Note: SNAP = Selected Nomenclature for Air Pollution.

9.3.1 Attribution of Damages to Source Sectors

First, the relation between different economic activities and environmental impacts was examined. For this purpose, emission scenarios were built by assuming a 100 per cent emission reduction in each individual CORINAIR main economic sector of Germany, Italy, the Netherlands and the UK. Additionally, four scenarios were assumed in which all anthropogenic emissions of each of these countries were set to zero. Finally, by applying the air quality models and estimating the different effects through the respective exposure to air pollutants, the impacts and damage costs caused by the respective economic sectors were derived (see also Section 6.3, Chapter 6). The 11 main sectors of CORINAIR 1990 are listed in Table 9.2. A detailed description of the sectors can be found in McInnes (1996).

9.3.2 Attribution of Damages to Countries' Emissions

A main characteristic of air pollution is the transboundary transport of pollutants which leads to an 'import' and 'export' of damages between countries through the atmosphere. The attribution of damages caused in the EU-15 states was analysed by estimating the geographical distribution of damages in Europe caused by each of these countries. Therefore, 15 emission scenarios, one for each country, were built. Each of the reduction scenarios includes only the natural emissions of the analysed country and all 1990 emissions of the remaining regions in the model area. The emissions of SO_2, NO_X, NH_3 and PM_{10} (the last for Germany only) were considered, while damages caused by ozone had to be left out in this part of the study because the source receptor matrices for ozone used in the air quality model SROM could not be applied for reductions of anthropogenic emissions larger than 70 per cent.

9.4 IMPACTS AND DAMAGE COSTS CAUSED BY THE ECONOMIC SECTORS OF GERMANY, ITALY, THE NETHERLANDS AND THE UK

In this section the damages resulting from airborne emissions of the CORINAIR main economic sectors (see Table 9.3) in Germany, Italy, the Netherlands and the UK and from the whole countries' emissions are estimated. Before the impacts and damage costs are individually presented for each impact category, the emissions of the economic sectors in the four countries are discussed.

9.4.1 Emissions of Substances Related to Air Pollution in Germany, Italy, the Netherlands and the UK

The emissions of air pollution related substances are the first indicators of the environmental impacts of economic activities. In Table 9.3 the emissions of the CORINAIR main economic sectors for 1990, which are used in the current study, are listed for the four countries Germany, Italy, the Netherlands and the UK. The database includes emissions of SO_2, NO_X, NH_3 and NMVOC, which are spatially distributed according to the European NUTS (nomenclature of territorial units for statistics) classification, for example, state, country and district.

The economic sectors in the CORINAIR database are classified from a technical point of view. They represent the economic processes which are the main contributors to airborne pollutants. 'N.A.' in Table 9.3 indicates that no emission data are available in the CORINAIR database.

For Germany the geographically distributed direct emissions of particles with a diameter smaller than 10 μm (PM_{10}), which are not included in the CORINAIR 1990 database, were roughly estimated for each economic sector by attributing an average German emission rate of total suspended matter (TSP) per person to the population distribution.

The pollutants, which are formed by the emitted compounds (Table 9.4) through chemical reactions, are small secondary particles like nitrates and sulphates as well as ozone. Moreover, the depositions of acidic substances and nitrogen are taken into account. The directly emitted particles included in the study are regarded as inert. As the reaction schemes for most

Table 9.3 Emissions of CORINAIR main sectors of Germany, Italy, the Netherlands and the UK

Pollutant	Emission (kilotons/annum)										
	Public power, cogeneration, district heating plants	Commercial, institutional and residential combustion plants	Industrial combustion	(Non-combustion) production processes	Extraction and distribution of fuels	Solvent use	Road transport	Other mobile sources and machinery	Waste treatment and disposal	Agriculture	All economic sectors
Germany											
SO_2	2300	590	2200	61	20	N.A.	74	30	N.A.	N.A.	5300
NO_X	420	110	510	21	2.3	N.A.	1600	280	N.A.	N.A.	3000
NH_3	N.A.	4.9e–1	N.A.	3.5	N.A.	N.A.	8.0	N.A.	N.A.	730	740
NMVOC	6.2	110	11	120	200	1200	1200	77	N.A.	N.A.	2900
PM_{10}	450	260	400	280	N.A.	N.A.	41	28	N.A.	N.A.	1500
Italy											
SO_2	770	82	570	100	N.A.	N.A.	100	48	4.3	N.A.	1700
NO_X	410	59	300	12	2.1e–1	N.A.	950	280	34	5.0e–1	2000
NH_3	1.4e–1	1.8e–3	7.1e–2	23	N.A.	N.A.	5.9e–1	1.2e–2	10	350	380
NMVOC	4.0	21	11	97	130	540	950	130	110	400	2400

The Netherlands											
SO_2	44	4.1	43	74	9.0e–2	3.0e–1	13	17	4.8	1.4	200
NO_X	76	31	40	61	1.6	9.0e–1	270	49	5.3	24	560
NH_3	N.A.	1.0	6.5e–3	3.9	N.A.	7.3e–3	4.0e–01	N.A.	1.8e–1	190	200
NMVOC	7.0e–1	17	1.2	77	1.3	150	180	23	3.1	5.1	460
UK											
SO_2	2700	210	700	18	N.A.	N.A.	63	65	N.A.	N.A.	3800
NO_X	780	120	230	8.9	65	N.A.	1400	180	12	N.A.	2800
NH_3	N.A.	N.A.	N.A.	15	N.A.	N.A.	N.A.	N.A.	4.1	450	470
NMVOC	13	41	3.1	300	440	750	980	26	48	N.A.	2600

Note: N.A. = no emission data available.

Table 9.4 Emitted compounds and pollutants formed by chemical reactions

Emitted substance	Directly related pollutants
SO_2	SO_2, sulphates, deposition of acidic substances
NO_X	NO_2, nitrates, deposition of acidic substances and nitrogen, O_3
NH_3	nitrates, sulphates, deposition of acidic substances and nitrogen
PM_{10}	PM_{10}
NMVOC	O_3

substances are non-linear, that is, competing reactions exist, further pollutants influence the formation of the diverse atmospheric compounds. This can especially be observed for the formation of secondary particles. For example, fewer sulphates are produced if the NO_X emissions increase strongly without any increase in NH_3 emissions, because in this situation less NH_3 is available for the reaction with SO_2 to build sulphates.

Another example of the influence of non-linear chemical reactions is that, although reactions of NMVOC with NO_X are needed to form O_3, an additional increase of NO_X at high NO_X levels leads to a decrease of ozone concentration. Therefore, concerning impacts caused by O_3, benefits to the environment were actually observed due to activities of economic sectors with a large emission ratio of NO_X/NMVOC. This is especially the case with the energy and transport sectors; however, the same sectors also emit high amounts of SO_2 and NO_X, so they make a major contribution to particle related damages.

9.4.2 Impacts on and Damage Costs to Human Health, Field Crops and Building Materials

In the following, in addition to the damage costs for the individual sectors, damages caused by all anthropogenic emissions of the respective countries are derived from an emission scenario in which these emissions are set to zero. An exception is the damage costs caused by O_3 concentration, because the SROM model can only be used to model small changes in anthropogenic NO_X and NMVOC emissions (see comment above in Section 9.3). Therefore, for ozone, instead of the damages caused by all sectors, the sum of damage costs caused by the individual sectors is presented in the tables. This aggregated value gives an underestimation of the damages caused by ozone at high NO_X concentration levels, because the effect of an increase in NO_X concentrations leading to ozone reduction, and thereby to a benefit, is double counted for the individual economic sectors.

The estimated impacts are discussed in detail for each impact category, that is, mortality, morbidity, field crops and building materials. They are then summarised for each impact category by monetary valuation of the individual physical impacts. Damage costs for the effects on mortality are derived in the respective subsection in which effects of mortality are discussed to investigate the influence of the different discount rates of 3 and 11 per cent used in the evaluation. The damage costs for the remaining impacts can easily be estimated individually by applying the respective monetary values reported in Chapter 5.

Mortality

The estimated number of years of life lost (YOLL) due to acute and chronic mortality caused by the emissions of the main economic sectors in the four countries Germany, Italy, the Netherlands and the UK is shown in Table 9.5. Effects of acute and chronic mortality were taken into account.

By far the highest loss of life expectancy is caused by nitrates and sulphates related to the emissions of NO_X, SO_2 and NH_3. The estimation of the mortality effects of these pollutants results in about 480 000 years of life lost per year due to German emissions, 230 000 due to the UK's, 220 000 due to Italian and 51 000 due to the Netherlands' emissions. The evaluation of impacts caused by directly emitted particles shows that about 160 000 additional years of life could be expected to be lost annually as a result of German particle emissions of PM_{10} in 1990. The chronic mortality effects represent 99 per cent of the loss of lifetime caused by these particles. Hence these are much more important than the acute mortality impacts, which are also caused by SO_2 and O_3 concentrations.

A comparison of the mortality effects caused by different economic sectors for the four countries shows that the sectors 'Public power, cogeneration, district heating plants', 'Commercial, institutional and residential combustion plants', 'Agriculture' and 'Road transport' in each country are among the economic sectors which cause the most damages. In all four countries, benefits are observed for O_3 related impacts in economic sectors with a high emission ratio of NO_X/NMVOC. For the UK and the Netherlands these are even higher than the damages in the remaining sectors, so that when summarising the impacts of the individual sectors benefits can be observed.

The reason for the small benefits observed for emissions in the road transport sector for Germany and the sector 'Extraction and distribution of fossil fuels' in the UK concerning sulphate aerosols are the large emissions of NO_X which react with NH_3 to build nitrates. As a consequence

Table 9.5 Estimated acute and chronic effects on mortality from O_3, SO_2, nitrates and sulphates caused by German, Italian, the Netherlands, and UK economic activities in 1990

Function	Impacts (year)										
	Public power, cogeneration, district heating plants	Commercial, institutional and residential combustion plants	Industrial combustion	(Non-combustion) production processes	Extraction and distribution of fuels	Solvent use	Road transport	Other mobile sources and machinery	Waste treatment and disposal	Agriculture	All economic sectors
German emissions											
O_3											
Acute YOLL	−670	1.0	−790	160	320	1900	−220	−330	N.A.	N.A.	350
SO_2											
Acute YOLL	4800	1200	4800	170	48	N.A.	170	62	N.A.	−3.0e–3	11 000
Nitrates											
YOLL	7900	2400	12 000	590	−69	N.A.	93 000	13 000	N.A.	67 000	200 000
Sulphates											
YOLL	110 000	28 000	110 000	3 800	1200	N.A.	−270	710	N.A.	33 000	280 000
PM_{10}											
YOLL	50 000	28 000	45 000	31 000	N.A.	N.A.	4 600	3 100	N.A.	N.A.	160 000
Italian emissions											
O_3											
Acute YOLL	−190	−6.6	−140	96	140	560	760	2.4	100	N.A.	1300
SO_2											
Acute YOLL	1400	170	860	130	−8.9e–7	N.A.	190	81	8.9	−2.7e–4	2 800

Nitrates											
'Chronic' YOLL	17 000	2800	13 000	1400	12	N.A.	53 000	15 000	2500	27 000	130 000
Sulphates											
'Chronic' YOLL	40 000	4400	26 000	4900	−3.7e–1	N.A.	3800	1900	440	12 000	91 000
Netherlands' emissions											
O_3											
Acute YOLL	−200	−58	−100	−49	−2.3	210	−390	−95	−9.4	−700	−700
SO_2											
Acute YOLL	130	10	120	210	0.22	0.75	32	44	14	3.2	570
Nitrates											
'Chronic' YOLL	2900	1400	1 300	2400	67	36	12 000	1900	190	16 000	37 000
Sulphates											
'Chronic' YOLL	2600	190	2400	4200	2.3	15	290	860	270	4000	14 000
UK emissions											
O_3											
Acute YOLL	−1 600	−200	−450	330	390	890	−1 400	−320	32	N.A.	−2 300
SO_2											
Acute YOLL	4900	400	1200	33	−3.7e–5	N.A.	100	76	−7.7e–06	−5.9e–4	6700
Nitrates											
'Chronic' YOLL	8500	2700	3 400	1000	1 800	N.A.	45 000	5000	560	19 000	97 000
Sulphates											
'Chronic' YOLL	98 000	7600	24 000	1200	−68	N.A.	320	1700	85	11 000	140 000

fewer sulphate aerosols are produced by including the respective emissions of the economic sectors than by excluding them. A similar effect is observed for the German sector ‘Extraction and distribution of fossil fuels’ for which negative damages through nitrates were recorded.

The conditions described are of course also reflected in the damage costs derived for different discount rates, which are listed in Tables 9.6 to 9.9. Comparing the four countries, the highest damage costs for a discount rate of 3 per cent are estimated for Germany (26 billion Euro p.a.)[1] followed by the UK (13 billion Euro p.a.), Italy (12 billion Euro p.a.), and the Netherlands (2.6 billion Euro p.a.). In Germany the main contribution to the damages costs came from the emissions of the economic sectors ‘Public power, cogeneration and district heating plants’ and ‘Industrial combustion’ (6.6 and 6.8 billion Euro p.a.),[2] which accounted for about 52 per cent of the damage costs. For the UK the sector of ‘Public power, cogeneration and district heating plants’ and for Italy the sectors ‘Public power, cogeneration and district heating plants’ and ‘Road transport’ dominate the estimates. In the Netherlands, the emissions related to agricultural activities (sector ‘Agriculture’) cause the largest share of damage costs.

At a discount rate of 11 per cent the damage costs for acute effects increase, while the costs for chronic effects decrease. The reason is that acute effects describe cases of death which are instantaneously caused, so the value of present years increases for higher discount rates (Chapter 5), while chronic effects include cases of death with a latency of several years, so that the lifetime lost in the population lies in the future, for which the monetary value decreases for higher discount rates. While the share of acute effects for particles increases from 4 to 25 per cent, the acute effects of the remaining pollutants are, at an 11 per cent discount rate, of the same order of magnitude as the chronic effects caused by particles. The total damage costs assessed were 14 billion Euro p.a.[3] for German emissions, 6.3 billion Euro p.a. for UK emissions, 5.9 billion Euro p.a. for Italian emissions and 1.1 billion Euro p.a. for the Netherlands’ emissions. While the shares of damage costs between the economic sectors change slightly, because the effects caused by SO_2 and O_3 and thereby the related emissions are weighted higher at an 11 per cent discount rate, the main contributing sectors remain the same.

Morbidity

The individual morbidity effects calculated for the emissions from the economic sectors in the four countries are shown in Tables 9.10 to 9.14. The attribution of the effects caused by SO_2, nitrate and sulphate concentrations to source sectors reflects the same pattern as the mortality effects discussed in the previous section. Thus, the main damaging economic

Table 9.6 Damage costs due to acute and chronic effects on mortality from O_3, SO_2, nitrates, sulphates and PM_{10} caused by German economic activities, estimated for discount rates of 3 and 11 per cent

Function	Damages (million Euro)										
	Public power, cogeneration, district heating plants	Commercial, institutional and residential combustion plants	Industrial combustion	(Non-combustion) production processes	Extraction and distribution of fuels	Solvent use	Road transport	Other mobile sources and machinery	Waste treatment and disposal	Agriculture	All economic sectors
Discount rate 3%											
O_3											
Acute YOLL	−98	1.2e–01	−120	23	47	270	−33	−48	N.A.	N.A.	51
SO_2											
Acute YOLL	710	180	710	25	7.1	N.A.	26	9.2	N.A.	−4.9e–04	1700
Nitrates											
YOLL (96% chronic)	400	120	600	30	−3.5	N.A.	4800	670	N.A.	3400	10 000
Sulphates											
YOLL (96% chronic)	5600	1400	5600	190	60	N.A.	−14	36	N.A.	1700	14 000
Subtotal	6600	1700	6800	270	110	270	4700	670	N.A.	5100	26 000
PM_{10}											
YOLL (96% chronic)	2600	1500	2300	1600	N.A.	N.A.	230	160	N.A.	N.A.	8300
Total damages	9200	3200	9100	1900	N.A.	N.A.	4900	830	N.A.	N.A.	34 000

Table 9.6 (continued)

Function	Damages (million Euro)										
	Public power, cogeneration, district heating plants	Commercial, institutional and residential combustion plants	Industrial combustion	(Non-combustion) production processes	Extraction and distribution of fuels	Solvent use	Road transport	Other mobile sources and machinery	Waste treatment and disposal	Agriculture	All economic sectors
Discount rate 11%											
O_3											
'Acute' YOLL	−210	2.5e–1	−240	49	100	580	−70	−100	N.A.	N.A.	110
SO_2											
'Acute' YOLL	1500	370	1500	53	15	N.A.	54	19	N.A.	−1.0e–3	3500
Nitrates											
YOLL (75% chronic)	170	51	250	13	−1.5	N.A.	2000	280	N.A.	1400	4300
Sulphates											
YOLL (75% chronic)	2300	590	2300	81	25	N.A.	−5.8	15	N.A.	700	5900
Subtotal	3800	1000	3800	190	140	580	2000	210	N.A.	2100	14 000
PM_{10}											
YOLL (75% chronic)	1100	600	950	660	N.A.	N.A.	97	66	N.A.	N.A.	3500
Total damages	4900	1600	4800	850	N.A.	N.A.	2100	280	N.A.	N.A.	18 000

Table 9.7 Damage costs due to acute and chronic effects on mortality from O_3, SO_2, nitrates, sulphates and PM_{10} caused by Italian economic activities, estimated for discount rates of 3 and 11 per cent

Function	Damages (million Euro)										
	Public power, cogeneration, district heating plants	Commercial, institutional and residential combustion plants	Industrial combustion	(Non-combustion) production processes	Extraction and distribution of fuels	Solvent use	Road transport	Other mobile sources and machinery	Waste treatment and disposal	Agriculture	All economic sectors
Discount rate 3%											
O_3											
Acute YOLL	−29	−9.8e–1	−20	14	21	83	110	3.6e–1	15	N.A.	200
SO_2											
Acute YOLL	200	25	130	20	−1.3e–7	N.A.	27	12	1.3	−4.0e–5	420
Nitrates											
YOLL (96% chronic)	850	140	660	73	6.0e–1	N.A.	2700	740	130	1400	6500
Sulphates											
YOLL (96% chronic)	2000	230	1 300	250	N.A.	N.A.	200	98	22	630	4600
Subtotal	3100	390	2100	360	21	83	3000	850	170	2000	12 000

Table 9.7 (continued)

Function	Damages (million Euro)										
	Public power, cogeneration, district heating plants	Commercial, institutional and residential combustion plants	Industrial combustion	(Non-combustion) production processes	Extraction and distribution of fuels	Solvent use	Road transport	Other mobile sources and machinery	Waste treatment and disposal	Agriculture	All economic sectors
Discount rate 11%											
O_3											
Acute YOLL	−60	−2.1	−42	30	43	180	240	7.5e–1	31	N.A.	410
SO_2											
Acute YOLL	430	53	270	41	−2.8e–7	N.A.	58	25	2.8	−8.4e–5	880
Nitrates											
YOLL (75% chronic)	350	59	270	30	3.0e–1	N.A.	1100	310	54	570	2700
Sulphates											
YOLL (75% chronic)	850	94	550	100	−8.0e–3	N.A.	82	41	9.3	260	1900
Subtotal	1600	200	1 100	210	44	180	1500	380	97	830	5900

Table 9.8 Damage costs due to acute and chronic effects on mortality from O_3, SO_2, nitrates and sulphates caused by the Netherlands' economic activities, estimated for discount rates of 3 per cent and 11 per cent

Function	Damages (million Euro)										
	Public power, cogeneration, district heating plants	Commercial, institutional and residential combustion plants	Industrial combustion	(Non-combustion) production processes	Extraction and distribution of fuels	Solvent use	Road transport	Other mobile sources and machinery	Waste treatment and disposal	Agriculture	All economic sectors
Discount rate 3%											
O_3											
Acute YOLL	−29	−8.5	−15	−7.2	−0.3	30	−58	−14	−1.4	0.0e+0	−100
SO_2											
Acute YOLL	20	1.5	18	30	3.2e–2	0.11	4.7	6.5	2	0.47	84
Nitrates											
YOLL (96% chronic)	150	69	68	120	3.4	1.8	630	100	9.6	820	1 900
Sulphates											
YOLL (96% chronic)	130	9.6	120	220	0.12	0.77	15	44	14	210	740
Subtotal	270	72	200	360	3.3	33	600	140	24	1 000	2 600

Table 9.8 (continued)

Function	Damages (million Euro)										
	Public power, cogeneration, district heating plants	Commercial, institutional and residential combustion plants	Industrial combustion	(Non-combustion) production processes	Extraction and distribution of fuels	Solvent use	Road transport	Other mobile sources and machinery	Waste treatment and disposal	Agriculture	All economic sectors
Discount rate 11%											
O_3											
Acute YOLL	−61	−18	−32	−15	−0.7	64	−120	−30	−2.9	0.0e+0	−220
SO_2											
Acute YOLL	42	3.2	38	64	6.8e–2	0.23	10	14	4.3	0.99	180
Nitrates											
YOLL (75% chronic)	61	29	28	51	1.4	0.76	260	42	4	340	790
Sulphates											
YOLL (75% chronic)	55	4	52	90	4.9e–2	0.32	6.1	18	5.7	86	310
Subtotal	96	18	86	190	0.84	65	160	44	11	430	1100

Table 9.9 Damage costs due to acute and chronic effects on mortality from O_3, SO_2, nitrates and sulphates caused by UK economic activities, estimated for discount rates of 3 and 11 per cent

Function	Damages (million Euro)										
	Public power, cogeneration, district heating plants	Commercial, institutional and residential combustion plants	Industrial combustion	(Non-combustion) production processes	Extraction and distribution of fuels	Solvent use	Road transport	Other mobile sources and machinery	Waste treatment and disposal	Agriculture	All economic sectors
Discount rate 3%											
O_3											
Acute	−230	−41	−94	70	82	190	−290	−67	6.7	N.A.	−380
YOLL											
SO_2											
Acute	710	58	180	4.9	−5.5e–6	N.A.	15	11	−1.1e–6	−8.7e–5	980
YOLL											
Nitrates											
YOLL (96% chronic)	440	140	170	53	95	N.A.	2300	260	29	970	4900
Sulphates											
YOLL (96% chronic)	5000	390	1200	60	−3.5	N.A.	16	85	4.3	550	7100
Subtotal	6000	540	1500	190	170	190	2000	290	40	1500	13000

Table 9.9 (continued)

Function	Damages (million Euro)										
	Public power, cogeneration, district heating plants	Commercial, institutional and residential combustion plants	Industrial combustion	(Non-combustion) production processes	Extraction and distribution of fuels	Solvent use	Road transport	Other mobile sources and machinery	Waste treatment and disposal	Agriculture	All economic sectors
Discount rate 11%											
O_3											
Acute YOLL	−480	−87	−200	150	170	390	−620	−140	14	N.A.	−800
SO_2											
Acute YOLL	1500	120	380	10	−1.2e–5	N.A.	32	24	−2.4e–6	−1.8e–4	2100
Nitrates											
YOLL (75% chronic)	180	58	72	22	39	N.A.	960	110	12	400	2100
Sulphates											
YOLL (75% chronic)	2100	160	510	25	−1.4	N.A.	6.8	35	1.8	230	3000
Subtotal	3300	260	760	200	210	390	380	25	28	630	6300

Table 9.10 Estimated chronic effects on morbidity from sulphates, nitrates and O_3 caused by German economic activities

Function	Impacts										
	Public power, cogeneration, district heating plants	Commercial, institutional and residential combustion plants	Industrial combustion	(Non-combustion) production processes	Extraction and distribution of fuels	Solvent use	Road transport	Other mobile sources and machinery	Waste treatment and disposal	Agriculture	All economic sectors
Sulphates and Nitrates											
Congestive heart failure in adults over 65 yrs. (cases)	1 300	320	1 300	47	12	N.A.	990	150	N.A.	1 100	5 100
Chronic bronchitis in adults (cases)	20 000	5 100	20 000	730	180	N.A.	16 000	2 400	N.A.	17 000	81 000
Restricted activity days in adults (1000*days)	10 000	2700	11 000	390	97	N.A.	8 200	1 200	N.A.	8 900	42 000
Bronchodilator usage for asthma (adults) (1000* cases)	1 700	440	1 700	63	16	N.A.	1 300	200	N.A.	1 400	6 900
Cough in adult asthmatics (1000*days)	1 700	450	1 800	65	16	N.A.	1 400	200	N.A.	1 500	7 100

Table 9.10 (continued)

Function	Impacts										
	Public power, cogeneration, district heating plants	Commercial, institutional and residential combustion plants	Industrial combustion	(Non-combustion) production processes	Extraction and distribution of fuels	Solvent use	Road transport	Other mobile sources and machinery	Waste treatment and disposal	Agriculture	All economic sectors
Lower respiratory symptoms in adult asthmatics. (1000*days)	630	160	650	23	5.9	N.A.	500	74	N.A.	540	2600
Bronchodilator usage in asthmatics (children) (1000*cases)	340	88	350	13	3.2	N.A.	270	40	N.A.	290	1400
Cough in asthmatics (children) (1000*days)	590	150	600	22	5.4	N.A.	460	69	N.A.	500	2400
Lower respiratory symptoms in asthmatics (children) (1000*days)	450	120	460	17	4.2	N.A.	360	53	N.A.	380	1800

Cases of child bronchitis (1000*cases)	200	52	210	7.5	1.9	N.A.	160	24	N.A.	170	820
Chronic cough in children (1000*episodes)	260	67	270	9.6	2.4	N.A.	200	30	N.A.	220	1 100
Cerebrovascular hospital admissions (cases)	2 600	680	2 700	98	25	N.A.	2 100	310	N.A.	2 200	11 000
All respiratory hospital admissions (cases)	1 100	280	1 100	40	10	N.A.	850	130	N.A.	920	4 400
O_3											
Asthma attacks in adults over 65 yrs. (cases)	−230	2.7e–1	−270	55	110	640	−77	−110	N.A.	N.A.	120
Minor restricted activity day in adults (1000* days)	−1 200	1.4	−1 400	280	580	3 300	−400	−580	N.A.	N.A.	620
All respiratory hospital admissions (1000*cases)	−540	6.5e–1	−640	130	260	1 500	−180	−260	N.A.	N.A.	280

Table 9.10 (continued)

Function	Impacts										
	Public power, cogeneration, district heating plants	Commercial, institutional and residential combustion plants	Industrial combustion	(Non-combustion) production processes	Extraction and distribution of fuels	Solvent use	Road transport	Other mobile sources and machinery	Waste treatment and disposal	Agriculture	All economic sectors
Symptom days in total population (1000*days)	−5000	6	−6000	1200	2400	14 000	−1700	−2500	N.A.	N.A.	2600
SO_2											
All respiratory hospital admissions (cases)	1800	460	1800	64	18	N.A.	66	24	N.A.	−1.1e–3	4300

Table 9.11 Estimated chronic effects on morbidity from PM_{10} caused by economic activities in Germany

Function	Impacts										
	Public power, cogeneration, district heating plants	Commercial, institutional and residential combustion plants	Industrial combustion	(Non-combustion) production processes	Extraction and distribution of fuels	Solvent use	Road transport	Other mobile sources and machinery	Waste treatment and disposal	Agriculture	All economic sectors
PM_{10}											
Congest. heart failure in adults over 65 yrs. (cases)	540	300	480	330	N.A.	N.A.	49	33	N.A.	N.A.	1700
Chronic bronchitis in adults (cases)	8800	5000	7800	5400	N.A.	N.A.	800	540	N.A.	N.A.	28000
Restricted activity days in adults (1000*days)	4500	2500	4000	2800	N.A.	N.A.	410	280	N.A.	N.A.	14000
Bronchodilator usage in asthmatic adults (1000*cases)	730	410	650	450	N.A.	N.A.	66	45	N.A.	N.A.	2300
Cough in asthmatic adults (1000*days)	750	420	670	460	N.A.	N.A.	68	46	N.A.	N.A.	2400
Lower respiratory symptoms in asthmatic adults (1000*days)	270	150	240	170	N.A.	N.A.	25	17	N.A.	N.A.	870

Table 9.11 (continued)

Function	Impacts									
	Public power, cogeneration, district heating plants	Commercial, institutional and residential combustion plants	Industrial combustion	(Non-combustion) production processes	Extraction and distribution of fuels	Solvent use	Road transport	Other mobile sources and machinery	Waste treatment and disposal	Agriculture	All economic sectors
Bronchodilator usage in asthamtic children (1000* cases)	150	82	130	90	N.A.	N.A.	13	9.0	N.A.	N.A.	470
Cough in asthmatic children (1000* days)	250	140	220	160	N.A.	N.A.	23	16	N.A.	N.A.	810
Lower respiratory symptoms in asthmatic children (1000*days)	190	110	170	120	N.A.	N.A.	18	12	N.A.	N.A.	620
Case of chronic bronchitis in children (1000* cases)	87	49	77	54	N.A.	N.A.	7.8	5.4	N.A.	N.A.	280
Chronic cough in children (1000* episodes)	110	63	99	69	N.A.	N.A.	10	6.9	N.A.	N.A.	360
Cerebrovascular hospital admissions (cases)	1100	640	1000	700	N.A.	N.A.	100	70	N.A.	N.A.	3 600
Respiratory hospital admissions in total pop. (cases)	460	260	410	290	N.A.	N.A.	42	29	N.A.	N.A.	1 500

Table 9.12 *Estimated chronic effects on morbidity from sulphates, nitrates, SO_2 and O_3 caused by economic activities in Italy*

Function	Impacts										
	Public power, cogeneration, district heating plants	Commercial, institutional and residential combustion plants	Industrial combustion	(Non-combustion) production processes	Extraction and distribution of fuels	Solvent use	Road transport	Other mobile sources and machinery	Waste treatment and disposal	Agriculture	All economic sectors
Sulphates and Nitrates											
Congestive heart failure in adults over 65 yrs. (cases)	600	77	420	68	1.2e–1	N.A.	600	180	32	420	2 300
Chronic bronchitis in adults (cases)	9500	1 200	6 600	1 100	2.0	N.A.	9800	2800	510	6 700	37 000
Restricted activity days in adults (1000*days)	5 000	640	3400	560	1.0	N.A.	5000	1500	260	3 400	19 000
Bronchodilator usage in asthmatic adults (1000*cases)	810	100	560	91	1.7e–1	N.A.	820	240	43	560	3100
Cough in asthmatic adults (1000*days)	840	110	580	94	1.7e–1	N.A.	840	240	44	580	3200
Lower respiratory symptoms in asthmatic adults (1000*days)	300	39	210	34	6.2e–2	N.A.	300	88	16	210	1200

Table 9.12 (continued)

Function	Impacts										
	Public power, cogeneration, district heating plants	Commercial, institutional and residential combustion plants	Industrial combustion	(Non-combustion) production processes	Extraction and distribution of fuels	Solvent use	Road transport	Other mobile sources and machinery	Waste treatment and disposal	Agriculture	All economic sectors
Bronchodilator usage in asthmatic children (1000* cases)	160	21	110	18	3.3e–2	N.A.	160	48	8.6	110	630
Cough in asthmatic children (1000* days)	280	36	190	32	5.8e–2	N.A.	280	82	15	190	1100
Lower respiratory symptoms in asthmatic children (1000*days)	220	28	150	24	4.4e–2	N.A.	220	63	11	150	830
Case of chronic bronchitis in children (1000* cases)	97	12	67	11	2.0e–2	N.A.	97	28	5.1	67	370
Chronic cough in children (1000* episodes)	120	16	86	14	2.6e–2	N.A.	120	36	6.5	86	480

Cerebrovascular hospital admissions (cases)	1300	160	870	140	2.6e–01	N.A.	1300	370	66	870	4900
All respiratory hospital admissions (cases)	520	66	360	58	1.1e–01	N.A.	520	150	27	360	2000
O_3											
Asthma attacks in adults over 65 yrs. (1000*cases)	−67	−2.3	−47	33	48	190	260	8.4e–1	35	N.A.	460
Minor restricted activity days in adults (1000*days)	−350	−12	−240	170	250	1000	1400	4.4	180	N.A.	2 400
All respiratory hospital admissions (cases)	−160	−5.4	−110	78	110	460	620	2.0	82	N.A.	1100
Symptom days in total population (1000*days)	−1 500	−50	−1000	730	1100	4300	5800	18	760	N.A.	10 000
SO_2											
Respiratory hospital admissions in total population (cases)	530	64	330	50	−3.4e–7	N.A.	70	31	3.4	−1.0e–4	1100

Table 9.13 Estimated chronic effects on morbidity from sulphates, nitrates, SO_2 and O_3 caused by the economic activities in the Netherlands

Function	Impacts										
	Public power, cogeneration, district heating plants	Commercial, institutional and residential combustion plants	Industrial combustion	(Non-combustion) production processes	Extraction and distribution of fuels	Solvent use	Road transport	Other mobile sources and machinery	Waste treatment and disposal	Agriculture	All economic sectors
Sulphates and Nitrates											
Congestive heart failure in adults over 65 yrs. (cases)	58	16	40	70	0.73	0.54	140	30	4.9	210	550
Chronic bronchitis in adults (cases)	930	270	630	1100	12	8.7	2200	480	77	3500	8900
Restricted activity days in adults (1000*days)	480	140	330	580	6.1	4.5	1100	250	40	1800	4600
Bronchodilator usage in asthmatic adults (1000*cases)	79	22	54	95	0.99	0.73	180	40	6.6	290	740

Cough in asthmatic adults (1000*days)	81	23	56	98	1.0	0.75	190	42	6.8	300	770
Lower respiratory symptoms in asthmatic adults (1000*days)	29	8.3	20	35	0.37	0.27	68	15	2.4	110	280
Bronchodilator usage in asthmatic children (1000*cases)	16	4.5	11	19	0.20	0.15	37	8.1	1.3	58	150
Cough in asthmatic children (1000*days)	27	7.7	19	33	0.34	0.25	63	14	2.3	100	260
Lower respiratory symptoms in asthmatic children (1000*days)	21	5.9	14	25	0.26	0.19	49	11	1.8	77	200
Cases of chronic bronchitis in children (1000*cases)	9.4	2.7	6.5	11	0.12	8.70E–02	22	4.8	0.78	35	89

Table 9.13 (continued)

Function	Impacts										
	Public power, cogeneration, district heating plants	Commercial, institutional and residential combustion plants	Industrial combustion	(Non-combustion) production processes	Extraction and distribution of fuels	Solvent use	Road transport	Other mobile sources and machinery	Waste treatment and disposal	Agriculture	All economic sectors
Chronic cough in children (1000*episodes)	12	3.4	8.3	15	0.15	0.11	28	6.2	1	44	110
Cerebrovascular hospital admissions (cases)	120	35	84	150	1.5	1.1	280	63	10	450	1200
All respiratory hospital admissions (cases)	50	14	35	61	0.63	0.47	120	26	4.2	180	480
O_3											
Asthma attacks in adults over 65 yrs. (1000*cases)	−68	−20	−35	−17	−0.8	71	−140	−33	−3.2	0.0e+0	−240
Minor restricted activity days in adults (1000*days)	−350	−100	−180	−87	−4.1	370	−710	−170	−17	0.0e+0	−1300

All respiratory hospital admissions (cases)	−160	−47	−83	−40	−1.9	170	−320	−77	−7.6	0.0e+0	−570
Symptom days in total population (1000*days)	−1500	−440	−770	−370	−17	1600	−3000	−720	−71	0.0e+0	−5300
SO_2											
Respiratory hospital admissions in total population (cases)	51	3.8	46	78	8.3e–2	0.28	12	17	5.2	1.2	210

Table 9.14 Estimated chronic effects on morbidity from sulphates, nitrates, SO_2 and O_3 caused by economic activities in the United Kingdom

Function	Impacts										
	Public power, cogeneration, district heating plants	Commercial, institutional and residential combustion plants	Industrial combustion	(Non-combustion) production processes	Extraction and distribution of fuels	Solvent use	Road transport	Other mobile sources and machinery	Waste treatment and disposal	Agriculture	All economic sectors
Sulphates and Nitrates											
Congestive heart failure in adults over 65 yrs. (cases)	1100	110	290	24	19	N.A.	490	72	6.9	320	2500
Chronic bronchitis in adults (cases)	18 000	1700	4600	380	310	N.A.	7900	1200	110	5100	40 000
Restricted activity days in adults (1000*days)	9500	910	2400	200	160	N.A.	4000	590	57	2600	21 000
Bronchodilator usage in asthmatic adults (1000*cases)	1500	150	400	32	26	N.A.	660	97	9.3	430	3400

Cough in asthmatic adults (1000*days)	1 600	150	410	33	26	N.A.	670	99	9.6	440	3 500
Lower respiratory symptoms in asthmatic adults (1000*days)	570	55	150	12	9.5	N.A.	240	36	3.5	160	1 300
Bronchodilator usage in asthmatic children (1000*cases)	310	30	79	6.4	5.1	N.A.	130	19	1.9	86	680
Cough in asthmatic children (1000*days)	530	51	140	11	8.9	N.A.	230	33	3.2	150	1 200
Lower respiratory symptoms in asthmatic children (1000*days)	410	40	110	8.5	6.8	N.A.	170	26	2.5	110	910
Case of chronic bronchitis in children (1000*cases)	180	18	47	3.8	3.1	N.A.	78	12	1.1	51	410

Table 9.14 (continued)

Function	Impacts										
	Public power, cogeneration, district heating plants	Commercial, institutional and residential combustion plants	Industrial combustion	(Non-combustion) production processes	Extraction and distribution of fuels	Solvent use	Road transport	Other mobile sources and machinery	Waste treatment and disposal	Agriculture	All economic sectors
Chronic cough in children (1000*episodes)	240	23	61	4.9	3.9	N.A.	100	15	1.4	66	520
Cerebrovascular hospital admissions (cases)	2 400	230	610	50	40	N.A.	1000	150	15	670	5300
All respiratory hospital admissions (cases)	980	95	250	20	16	N.A.	420	62	6	270	2200
O_3											
Asthma attacks in adults over 65 yrs. (1000*cases)	−540	−68	−160	110	140	310	−480	−110	11	N.A.	−780
Minor restricted activity days in adults (1000*days)	−2800	−350	−810	590	700	1600	−2500	−570	57	N.A.	−4100

All respiratory hospital admissions (cases)	−1300	−160	−370	270	320	720	−1100	−260	26	N.A.	−1800
Symptom days in total population (1000*days)	−12 000	−1500	−3400	2500	3000	6700	−11 000	−2400	240	N.A.	−17 000
SO_2											
Respiratory hospital admissions in total pop. (cases)	1800	150	460	13	−1.4e–5	N.A.	38	29	−2.9e–6	−2.2e–4	2500

sectors remain 'Public power, cogeneration, and district heating plants', 'Industrial combustion', 'Road transport', and 'Agriculture'. Similar to the mortality effects, benefits are observed for some sectors concerning O_3 related impacts. In the case of the Netherlands and the UK these are dominating the morbidity effects due to O_3.

The damage costs estimated for the morbidity impacts in 1990 are about 24 billion Euro p.a.[4] for Germany, 14 billion Euro p.a. for the UK, 11 billion Euro p.a. for Italy and 2.5 billion Euro p.a. for the Netherlands. These figures are of the same order of magnitude as the damage costs assessed for mortality effects at a 3 per cent discount rate. In each country the main part of the morbidity costs (about 80 per cent) is due to chronic bronchitis.

Crops

Impacts on field crops caused by German, Italian, the Netherlands' and UK emissions are shown in Tables 9.15 to 9.18. The amount of limestone required for the neutralisation of acid deposition is estimated at 2000 kilotons p.a. due to emissions from Germany and the UK respectively, 1500 kilotons p.a. due to emissions from Italy and 180 kilotons p.a. due to emissions from the Netherlands. For nitrogen deposition only the fertilising effect is included in the study, while the effect of eutrophication was not taken into account. The savings on fertiliser in agriculture due to increased nitrogen deposition is assessed at 240 kilotons p.a. for Italian and UK emissions respectively, 200 kilotons p.a. for German emissions, and 62 kilotons p.a. for the Netherlands' emissions.

The decrease of O_3 caused in some regions by an increase of NO_X emissions at high NO_X levels has also had an effect on the AOT40 (Accumulated Ozone Concentrations Above a Threshold of 40 ppbV) values, which were derived for crops damages. Indeed, the situation for crops is more difficult than for human health damages, because the different crop species show different spatial distributions over Europe. Therefore the signs of the derived impact depends to a great extent on the regions in which O_3 increases and decreases. Thus, for some economic sectors with a high ratio of NO_X/NMVOC emissions benefits were estimated, but not for all. While the crop losses due to the individual sector emissions are higher than the benefits for all crop species in Germany and Italy, the results for the emissions of the UK and the Netherlands are dominated by positive yield changes.

The impacts of SO_2 concentrations on crops have a fertilising effect at low concentration levels and a damaging effect at high concentration levels. As the SO_2 concentration is comparatively low in large parts of Europe, benefits are observed for most crop species due to the emissions from the

Table 9.15 Estimated impacts on crops of effects of acid deposition, nitrogen deposition, O_3 and SO_2 resulting from German emissions

Function	Impacts										
	Public power, cogeneration, district heating plants	Commercial, institutional and residential combustion plants	Industrial combustion	(Non-combustion) production processes	Extraction and distribution of fuels	Solvent use	Road transport	Other mobile sources and machinery	Waste treatment and disposal	Agriculture	All economic sectors
Acid deposition											
Add. lime needed (kt) for all crops	650	150	570	17	5	N.A.	260	46	N.A.	310	2 000
Nitrogen deposition											
Add. fertiliser needed (kt) for all crops	−19	−4.8	−22	−1.2	−1.0e–1	N.A.	−68	−11	N.A.	−86	−200
O_3											
Yield loss (kt) of barley	−160	56	−190	99	180	1000	760	−42	N.A.	N.A.	1700
Yield loss (kt) of oats	−9.4	3.8	−11	6.4	12	66	51	−2.2	N.A.	N.A.	120
Yield loss (kt) of potato	−470	110	−550	230	430	2500	1500	−160	N.A.	N.A.	3500
Yield loss (kt) of rice	−8.2e–1	5.0e–1	−9.6e–1	7.3e–1	1.3	7.5	6.4	−8.0e–2	N.A.	N.A.	15
Yield loss (kt) of rye	−13	5.4	−16	9	16	94	75	−3	N.A.	N.A.	170

Table 9.15 (continued)

Function	Impacts										
	Public power, cogeneration, district heating plants	Commercial, institutional and residential combustion plants	Industrial combustion	(Non-combustion) production processes	Extraction and distribution of fuels	Solvent use	Road transport	Other mobile sources and machinery	Waste treatment and disposal	Agriculture	All economic sectors
Yield loss (kt) of sunflower seed	−9.3	4.4	−11	7	13	72	59	−1.7	N.A.	N.A.	130
Yield loss (kt) of tobacco	−7.2e–1	7.4e–1	−8.3e–1	9.7e–1	1.7	9.8	9.7	1.3e–1	N.A.	N.A.	21
Yield loss (kt) of wheat	−420	130	−490	250	460	2700	1800	−130	N.A.	N.A.	4200
SO_2											
Yield loss (kt) of barley	410	100	410	11	1.3	N.A.	4.6	1.7	N.A.	−5.7e–5	950
Yield loss (kt) of oats	47	11	43	1.5	1.5e–1	N.A.	5.0e–1	1.8e–1	N.A.	−5.0e–6	110
Yield loss (kt) of potato	990	230	910	25	2.7	N.A.	8.9	3.2	N.A.	−6.0e–5	2200
Yield loss (kt) of rye	140	28	110	2.5	2.3e–1	N.A.	7.9e–1	2.9e–1	N.A.	−5.6e–6	280
Yield loss (kt) of sugar beet	600	140	540	16	2.7	N.A.	9.1	3.4	N.A.	−6.2e–5	1300
Yield loss (kt) of wheat	640	160	610	17	2.5	N.A.	9.0	3.4	N.A.	−1.4e–4	1500

Table 9.16 Estimated impacts on crops of effects of acid deposition, nitrogen deposition, O_3, and SO_2 resulting from Italian emissions

Function	Impacts										
	Public power, cogeneration, district heating plants	Commercial, institutional and residential combustion plants	Industrial combustion	(Non-combustion) production processes	Extraction and distribution of fuels	Solvent use	Road transport	Other mobile sources and machinery	Waste treatment and disposal	Agriculture	All economic sectors
Acid deposition											
Add.lime needed (kt) for all crops	390	48	280	57	4.9e–2	N.A.	260	83	19	320	1500
Nitrogen deposition											
Add. fertiliser needed (kt) for all crops	−26	−3.8	−19	−6.3	−1.4e–2	N.A.	−62	−18	−4.9	−91	−240
O_3											
Yield loss (kt) of barley	4.2	4.7	4.7	20	27	110	240	28	23	N.A.	460
Yield loss (kt) of oats	0.57	0.41	0.55	1.6	2.1	8.6	20	2.4	1.8	N.A.	38
Yield loss (kt) of potato	5.1	7.0	6.3	30	41	170	370	43	35	N.A.	710
Yield loss (kt) of rice	−6.0	−0.28	−4.3	2.7	3.9	16	22	−0.36	2.8	N.A.	36
Yield loss (kt) of rye	0.42	0.19	0.36	0.62	0.84	3.4	7.9	1.1	0.73	N.A.	15

Table 9.16 (continued)

Function	Impacts										
	Public power, cogeneration, district heating plants	Commercial, institutional and residential combustion plants	Industrial combustion	(Non-combustion) production processes	Extraction and distribution of fuels	Solvent use	Road transport	Other mobile sources and machinery	Waste treatment and disposal	Agriculture	All economic sectors
Yield loss (kt) of sunflower seed	3.4	1.1	2.8	2.9	3.8	15	41	6	3.50	N.A.	80
Yield loss (kt) of tobacco	−1.7	0.50	−0.98	3.6	4.9	20	42	3.6	4.0	N.A.	75
Yield loss (kt) of wheat	29	22	29	85	110	460	1100	130	98	N.A.	2100
SO_2											
Yield loss (kt) of barley	−6.7	−0.54	−3.8	−0.48	1.5e–8	N.A.	−0.66	−0.28	−2.2e–2	−4.6e–6	−17
Yield loss (kt) of oats	−1.4	−0.12	−1.1	−0.15	1.5e–9	N.A.	−0.18	−7.7e–2	−4.4e–3	2.0e–7	−3.4
Yield loss (kt) of potato	7.0	1.20	2.70	0.55	−1.3e–8	N.A.	0.80	0.37	7.7e–2	−7.4e–6	8.9
Yield loss (kt) of rye	2.2	0.29	1.2	0.18	1.3e–9	N.A.	0.29	0.13	1.6e–2	−9.7e–7	4.3
Yield loss (kt) of sugar beet	−46	−2.8	−18	−2.3	−7.3e–8	N.A.	−3.7	−1.6	−0.15	−8.9e–6	−100
Yield loss (kt) of wheat	−37	−2.8	−21	−2.8	−5.0e–8	N.A.	−4.2	−1.8	−0.13	−2.6e–5	−84

Table 9.17 Estimated impacts on crops of effects of acid deposition, nitrogen deposition, O_3, and SO_2 resulting from Netherlands' emissions

Function	Impacts										
	Public power, cogeneration, district heating plants	Commercial, institutional and residential combustion plants	Industrial combustion	(Non-combustion) production processes	Extraction and distribution of fuels	Solvent use	Road transport	Other mobile sources and machinery	Waste treatment and disposal	Agriculture	All economic sectors
Acid deposition											
Add. lime needed (kt) for all crops	24	6.4	18	32	0.26	0.23	45	13	2.3	130	280
Nitrogen deposition											
Add. fertiliser needed (kt) for all crops	−3.2	−1.5	−1.7	−3.2	−6.5e–2	−3.9e–2	−11	−2.1	−0.26	−37	−62
O_3											
Yield loss (kt) of barley	−60	−13	−31	4.7	−0.37	99	−48	−23	−2.1	0.0e+0	−74
Yield loss (kt) of oats	−3.7	−0.81	−1.9	0.31	−2.0e–2	6.1	−3.0	−1.4	−0.13	0.0e+0	−4.5
Yield loss (kt) of potato	−180	−47	−94	−15	−1.6	250	−260	−80	−7.5	0.0e+0	−440
Yield loss (kt) of rice	−0.19	−6.7e–3	−0.1	0.19	1.7e–03	0.65	0.39	−3.0e–2	1.7e–4	0.0e+0	0.91
Yield loss (kt) of rye	−5.0	−1.1	−2.6	0.33	−3.0e–2	8.1	−3.9	−1.9	−0.17	0.0e+0	−6.2

Table 9.17 (continued)

Function	Impacts										
	Public power, cogeneration, district heating plants	Commercial, institutional and residential combustion plants	Industrial combustion	(Non-combustion) production processes	Extraction and distribution of fuels	Solvent use	Road transport	Other mobile sources and machinery	Waste treatment and disposal	Agriculture	All economic sectors
Yield loss (kt) of sunflower seed	−4	−0.75	−2	1.0	−1.0e–2	8.0	−0.81	−1.4	−0.11	0.0e+0	−0.11
Yield loss (kt) of tobacco	−0.28	−2.0e–2	−0.14	0.20	1.3e–3	0.79	0.30	−6.0e–2	−2.4e–3	0.0e+0	0.79
Yield loss (kt) of wheat	−170	−40	−87	5.2	−1.2	290	−170	−68	−6.2	0.0e+0	−250
SO_2											
Yield loss (kt) of barley	−0.34	−3.1e–2	−0.36	−0.63	−7.2e–4	−2.3e–3	−9.8e–2	−0.13	−3.7e–2	−1.1e–2	−1.9
Yield loss (kt) of oats	−8.9e–2	−7.7e–3	−8.4e–2	−0.14	−1.7e–4	−5.3e–4	−2.5e–2	−2.9e–2	−8.6e–3	−2.7e–3	−0.40
Yield loss (kt) of potato	−1200	−0.11	−1 200	−2400	−2.4e–3	−7.7e–3	−0.37	−0.46	−0.13	−4.0e–2	−8.5
Yield loss (kt) of rye	9.3e–2	6.1e–3	7.4e–2	0.12	1.1e–4	4.7e–4	1.9e–2	2.7e–2	7.9e–3	1.9e–3	0.30
Yield loss (kt) of sugar beet	−4000	−0.3	−3500	−6600	−6.7e–3	−2.5e–2	−0.98	−1500	−0.39	−0.10	−20
Yield loss (kt) of wheat	−1700	−0.13	−1700	−3000	−3.0e–3	−1.2e–2	−0.41	−0.70	−0.18	−4.2e–2	−8.5

Table 9.18 Estimated impacts on crops of effects of acid deposition, nitrogen deposition, O_3 and SO_2 resulting from UK emissions

Function	Impacts										
	Public power, cogeneration, district heating plants	Commercial, institutional and residential combustion plants	Industrial combustion	(Non-combustion) production processes	Extraction and distribution of fuels	Solvent use	Road transport	Other mobile sources and machinery	Waste treatment and disposal	Agriculture	All economic sectors
Acid deposition											
Add. lime needed (kt) for all crops	920	91	260	20	9.2	N.A.	230	45	5.3	370	2000
Nitrogen deposition											
Add. fert. needed (kt) for all crops	−31	−5.3	−9.1	−3.8	−2.6	N.A.	−60	−7.1	−1.5	−100	−240
O_3											
Yield loss (kt) of barley	−400	−45	−120	130	160	330	−130	−78	15	N.A.	−130
Yield loss (kt) of oats	−21	−2.3	−6.3	7.7	10	20	−3.2	−4.2	0.94	N.A.	1.5
Yield loss (kt) of potato	−820	−93	−240	260	330	690	−270	−160	31	N.A.	−280
Yield loss (kt) of rice	−0.36	1.7e–2	−0.13	0.65	0.94	1.7	1.8	−4.9e–2	9.9e–2	N.A.	4.7
Yield loss (kt) of rye	−19	−2.1	−6.1	9.2	12	24	5	−3.9	1.2	N.A.	20

Table 9.18 (continued)

Function	Impacts										
	Public power, cogeneration, district heating plants	Commercial, institutional and residential combustion plants	Industrial combustion	(Non-combustion) production processes	Extraction and distribution of fuels	Solvent use	Road transport	Other mobile sources and machinery	Waste treatment and disposal	Agriculture	All economic sectors
Yield loss (kt) of sunflower seed	−18	−1.7	−5.7	9.9	14	26	11	−3.6	1.3	N.A.	33
Yield loss (kt) of tobacco	−1.1	−6.6e–2	−0.37	0.97	1.4	2.5	1.9	−0.2	0.14	N.A.	5.1
Yield loss (kt) of wheat	−1300	−150	−370	370	470	990	−640	−250	41	N.A.	−850
SO_2											
Yield loss (kt) of barley	−35	0.18	0.26	0.16	−6.9e–7	N.A.	−3.3e–2	−0.52	1.3e–8	1.2e–6	−85
Yield loss (kt) of oats	−5.3	−0.20	−0.75	−8.0e–3	−6.0e–9	N.A.	−6.9e–2	−0.11	4.7e–9	2.6e–6	−9.8
Yield loss (kt) of potato	−45	−0.75	−3.6	4.3e–2	−4.7e–6	N.A.	−0.33	−0.77	−1.9e–6	−1.7e–5	−93
Yield loss (kt) of rye	0.29	7.0e–2	0.19	6.0e–3	−7.0e–7	N.A.	1.8e–0	1.2e–2	−2.4e–7	−2.0e–6	5.5e–2
Yield loss (kt) of sugar beet	−110	−4.7	−16	−0.23	−1.9e–6	N.A.	−1.40	−1.7	−9.7e–7	−3.0e–6	−190
Yield loss (kt) of wheat	−80	2.1	7.6	0.57	−2.6e–6	N.A.	0.69	−0.37	−3.3e–7	−9.2e–6	−210

UK, Italy and the Netherlands. Crop losses were only assessed for German emissions due to the high SO_2 level in Germany and especially in its Eastern neighbour countries. The sums of the yield changes caused by the individual sectors' emissions show large differences compared to the effects estimated by assuming emission changes in all economic sectors at the same time. This shows quite plainly the non-linear relation between impacts of the individual sectors and the damages caused by all sector emissions.

Some very small positive yield changes were assessed for emissions in the Agriculture sector in the UK and Germany in which only NH_3 is emitted. This can be explained by the reaction of SO_2 with NH_3 to form sulphates, which slightly decrease the concentration of SO_2.

A comparison of the absolute amounts of yield changes in all four countries for O_3 and SO_2 shows that O_3 is more important for crop damages than SO_2. Overall, the benefits estimated from the crop impacts of SO_2 and O_3 for the UK and the Netherlands are large enough to be dominant in the total crop damage costs so that benefits of 0.22 and 0.098 billion Euro p.a. are derived respectively (see Table 9.22). The damage costs assessed for the German and Italian emissions are 1.1 and 0.47 billion Euro p.a. These are at least one order of magnitude smaller than the mortality and morbidity effects on human health.

Material

The surfaces of materials that have to be maintained because of damages due to air pollution are presented in Tables 9.19 to 9.21. The largest area requiring maintenance work consisted of painted surfaces, which amounted to 110 km^2 (Germany), followed by 34 km^2 (Italy), 22 km^2 (UK) and 9 km^2 (the Netherlands). Other areas are given in the tables below.

The impacts on building materials show a similar attribution to the economic sectors as the human health impacts caused by SO_2, nitrate and sulphate aerosols. The reason is that one important emission in these sectors is SO_2, and acidic substances are intermediate products of the reactions, which lead to particle formation.

Damage costs resulting from effects on building materials amount to 1.7 billion Euro p.a. for German emissions, 1.4 billion Euro p.a. for UK emissions, 0.52 billion Euro p.a. for Italian emissions, and 0.14 billion Euro p.a. for the Netherlands' emissions. They are of the same order of magnitude as the crop damages, but about one order of magnitude smaller than the effects on human health (Table 9.22).

Summary and discussion of the results

Table 9.22 shows the damage costs caused by the individual economic sectors of the four countries, Germany, Italy, the Netherlands and the UK,

Table 9.19 Estimated surfaces that need to be maintained per year as an effect of acid deposition and SO_2 on building material caused by German emissions

Function	Maintenance surface										
	Public power, cogeneration, district heating plants	Commercial, institutional and residential combustion plants	Industrial combustion	(Non-combustion) production processes	Extraction and distribution of fuels	Solvent use	Road transport	Other mobile sources and machinery	Waste treatment and disposal	Agriculture	All economic sectors
German emissions											
Galvanised steel (1000*m^2)	2 900	670	2 600	82	17	N.A.	440	78	N.A.	690	8 000
Limestone (m^2)	4 000	1 200	4 800	230	10	N.A.	310	49	N.A.	490	14 000
Mortar (m^2)	32 000	8 700	33 000	1 300	330	N.A.	8 400	1 500	N.A.	11 000	94 000
Natural stone (m^2)	3 900	1 200	4 700	230	10	N.A.	260	42	N.A.	410	13 000
Paint (1000*m^2)	32 000	7 600	30 000	1 100	160	N.A.	8 100	1 300	N.A.	14 000	11 0000
Rendering (1000*m^2)	690	170	690	27	3.1	N.A.	81	14	N.A.	130	2 000
Sandstone (m^2)	5 800	1 700	6 900	330	15	N.A.	380	61	N.A.	590	19 000
Zinc (1000*m^2)	290	71	280	9.7	1.8	N.A.	44	6.9	N.A.	86	880

Table 9.20 Estimated surfaces which have to be maintained as an effect of acid deposition and SO_2 on building material caused by Italian and Netherlands' emissions

Function	Maintenance surface										
	Public power, cogeneration, district heating plants	Commercial, institutional and residential combustion plants	Industrial combustion	(Non-combustion) production processes	Extraction and distribution of fuels	Solvent use	Road transport	Other mobile sources and machinery	Waste treatment and disposal	Agriculture	All economic sectors
Italian emissions											
Galvanised steel ($1000*m^2$)	570	69	360	62	5.3e–2	N.A.	290	92	18	240	1 700
Limestone (m^2)	1400	200	930	170	1.2e–1	N.A.	670	210	48	890	4 500
Mortar (m^2)	4600	630	3100	450	3.8e–1	N.A.	2000	640	110	1200	12 000
Natural stone (m^2)	1400	190	890	160	9.8e–2	N.A.	580	180	41	730	4 100
Paint ($1000*m^2$)	9400	1200	6100	1200	1.4e+0	N.A.	7200	2100	470	7500	34 000
Rendering ($1000*m^2$)	190	24	120	21	1.5e–2	N.A.	87	28	5.6	87	560
Sandstone (m^2)	2000	270	1300	230	1.4e–1	N.A.	850	270	59	1100	6 000
Zinc ($1000*m^2$)	100	15	62	12	1.2e–2	N.A.	58	18	4.0	65	330

Table 9.20 (continued)

Function	Maintenance surface										
	Public power, cogeneration, district heating plants	Commercial, institutional and residential combustion plants	Industrial combustion	(Non-combustion) production processes	Extraction and distribution of fuels	Solvent use	Road transport	Other mobile sources and machinery	Waste treatment and disposal	Agriculture	All economic sectors
Netherlands' emissions											
Galvanised steel ($1000*m^2$)	73	14	63	110	0.51	0.56	87	31	7.4	230	600
Limestone (m^2)	200	31	170	290	1.0	1.4	190	75	19	650	1600
Mortar (m^2)	2400	410	2400	4600	14	21	2200	1100	300	7400	21 000
Natural stone (m^2)	200	28	160	280	0.90	1.3	160	71	18	540	1400
Paint ($1000*m^2$)	840	200	630	1100	7.8	6.8	1400	370	74	4600	9000
Rendering ($1000*m^2$)	21	3.2	17	30	0.11	0.14	20	7.9	2.0	59	160
Sandstone (m^2)	280	40	240	400	1.3	1.9	240	100	27	780	2100
Zinc ($1000*m^2$)	6.1	0.94	5.1	8.9	3.20E–02	4.20E–02	5.7	2.3	0.59	18	47

Table 9.21 *Estimated surfaces which have to be maintained as an effect of acid deposition and SO_2 on building material caused by UK emissions*

Function	Maintenance surface										
	Public power, cogeneration, district heating plants	Commercial, institutional and residential combustion plants	Industrial combustion	(Non-combustion) production processes	Extraction and distribution of fuels	Solvent use	Road transport	Other mobile sources and machinery	Waste treatment and disposal	Agriculture	All economic sectors
UK's emissions											
Galvanised steel (1000*m^2)	13 000	1100	3600	160	28	N.A.	920	280	17	1 200	21 000
Limestone (m^2)	1700	140	400	21	9.4	N.A.	280	53	4.6e+0	200	2 900
Mortar (m^2)	1 610 000	120 000	400 000	18 000	2600	N.A.	89 000	29 000	1700	130 000	2 390 000
Natural stone (m^2)	1700	140	380	19	7.7	N.A.	240	48	3.8	160	2 700
Paint (1000*m^2)	12 000	1000	2 800	200	99	N.A.	2700	450	49	2300	22 000
Rendering (1000*m^2)	640	47	140	6.5	1.7	N.A.	53	13	9.1e−1	54	960
Sandstone (m^2)	2400	200	560	28	11	N.A.	350	70	5.5	240	4000
Zinc (1000*m^2)	58	4.7	14	6.9e−1	2.7e−1	N.A.	7.9e+0	1.7	1.4e−1	6.2	95

Table 9.22 Estimated damages caused by German, Italian, the Netherlands' and UK economic activities

Receptor	Damages (million Euro)										
	Public power, cogeneration, district heating plants	Commercial, institutional and residential combustion plants	Industrial combustion	(Non-combustion) production processes	Extraction and distribution of fuels	Solvent use	Road transport	Other mobile sources and machinery	Waste treatment and disposal	Agriculture	All economic sectors
German emissions											
Material	510	120	490	17	2.7	N.A.	120	19	N.A.	200	1700
Crops	9.3	55	−43	61	110	640	400	−34	N.A.	−37	1100
Morbidity	5700	1500	5800	230	82	170	4700	660	N.A.	5000	24 000
Mortality (3% discount rate)	6600	1700	6800	270	110	270	4700	670	N.A.	5100	26 000
Total damages	13 000	3400	13 000	580	310	1100	9900	1300	N.A.	10 000	53 000
Damage export (%)	64	59	58	52	60	64	54	56	N.A.	44	58
Italian emissions											
Material	150	19	96	18	2.0e−2	N.A.	100	32	6.8	110	520
Crops	−14	3.7	−6.9	23	35	140	290	25	27	−39	470
Morbidity	2700	350	1900	320	13	50	2900	820	160	1900	11 000
Mortality (3% discount rate)	3100	390	2100	360	21	83	3000	850	15	2000	12 000
Total damages	5900	770	4100	720	69	280	6300	1700	210	4000	24 000
Damage export (%)	61	65	63	53	34	34	62	64	54	41	59

Netherlands' emissions											
Material	14	3.1	11	20	0.12	0.11	22	6.2	1.3	69	140
Crops	−43	−10	−22	−1.5	−0.33	65	−49	−18	−1.7	−16	−98
Morbidity	250	72	180	320	3.3	21	600	130	22	1000	2500
Mortality (3% discount rate)	270	72	200	360	3.2	30	600	140	−1.4	1000	2600
Total damages	490	140	360	700	6.3	120	1200	260	20	2100	5200
Damage export (%)	81	85	79	79	87	92	87	80	79	80	83
UK emissions											
Material	890	75	230	12	2.7	N.A.	83	21	1.5	96	1400
Crops	−270	−29	−71	76	98	210	−120	−51	8.4	−45	−220
Morbidity	5100	490	1300	140	120	79	2200	310	35	1500	11 000
Mortality (3% discount rate)	6000	540	1500	190	170	190	2000	290	6.7	1500	13 000
Total damages	12 000	1100	2900	410	400	470	4200	560	52	3000	25 000
Damage export (%)	50	48	46	56	66	76	54	47	61	33	52

classified by impact categories. Damage costs caused by German PM_{10} emissions are left out of the summary because no similar calculations were carried out for the remaining countries. As an example of the damage distribution by impact category, the damage costs of German emissions are shown in Figure 9.1. The effects on human health clearly dominate the damage costs. The main contributors to the damage costs are the economic sectors 'Public power, cogeneration and district heating plants', 'Industrial combustion plants', 'Road transport' and 'Agriculture'. Similar situations are observed for the emissions of the remaining three countries (Table 9.22).

For Germany, Italy and the UK all damage costs estimated for the emissions of the sector 'Solvent use' are related to ozone concentrations. For German emissions these represent at the same time the largest part of all damages caused by O_3 so that it can be easily seen in Table 9.22 and Figure 9.1 that O_3 related damages are not very important in the total amount of damage costs.

Figures 9.1 to 9.5 show the attribution of damage costs caused by the four countries Germany, Italy, the Netherlands and the UK inside and outside the respective countries to economic sectors. For all four countries, each of the sectors 'Public power, cogeneration, distinct heating plants', 'Commercial, institutional and residential combustion plants', 'Industrial combustion', 'Road transport' and 'Agriculture' are the five main contributors to the damage costs. Differences can be observed, especially in the ranking and the relative contribution to the damage costs of these four economic sectors. In the UK, for example, the highest damage costs of 12 billion Euro p.a. (46 per cent of total damage costs) are caused by the sector 'Public power, cogeneration and district heating plants', while in the Netherlands 2.1 billion Euro p.a. (46 per cent of total damage costs) are related to emissions of the sector 'Agriculture'.

Besides the total damage costs, Figures 9.2 to 9.5 and Table 9.22 show the share of damages caused inside and outside the countries ('damage export'). The share of damages occurring outside the country shows high deviations from the average value for some sectors. This is caused by different emission structures and geographical locations of the sources. The share of the damages inside the country is the largest for the UK and the smallest for the Netherlands. The reason is that most of the damages caused occur in an area of some 100 kilometres around the emission sources. Therefore, most impacts caused by the emissions of small countries like the Netherlands occur in neighbouring countries whilst most impacts due to emissions from island countries like the UK occur inside the country. A further analysis of damage distributions from several countries' emissions is carried out in the next section.

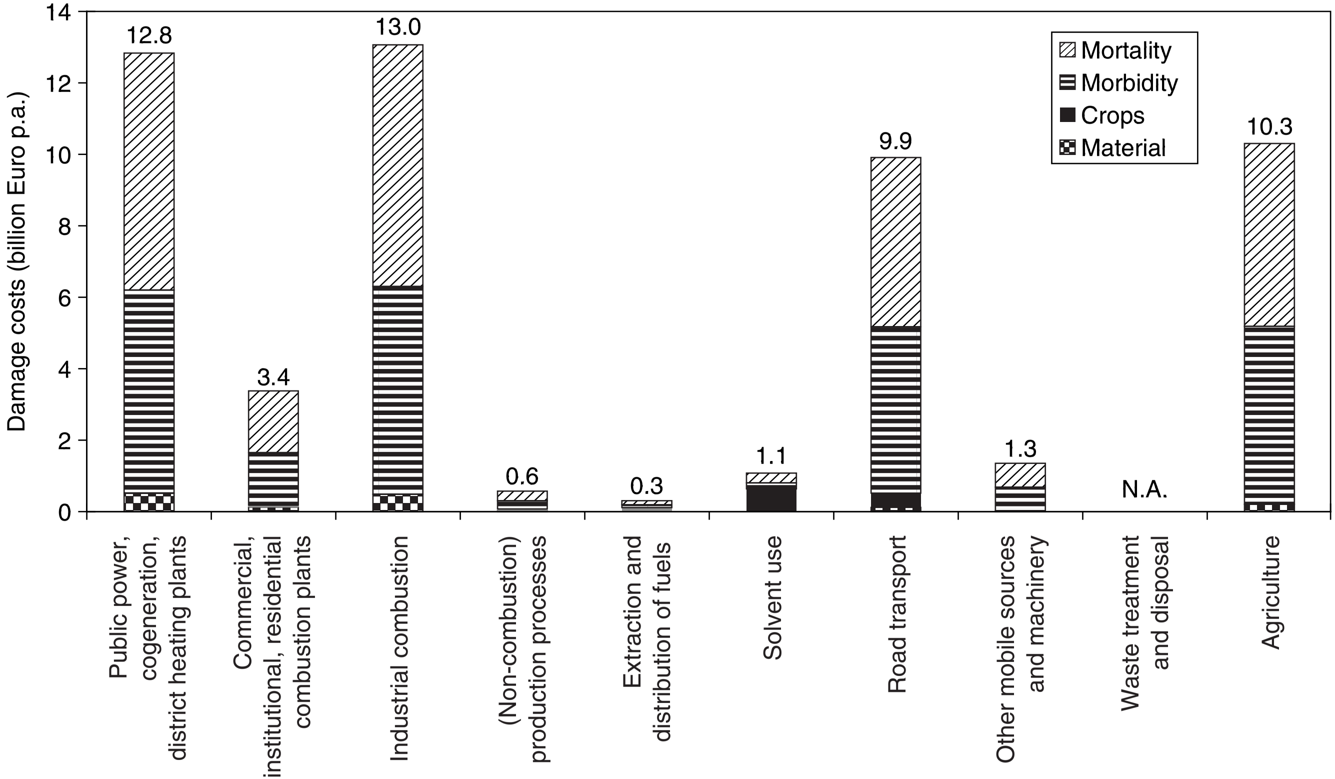

Figure 9.1 Damage costs caused by German economic sectors and their attribution to different impact categories

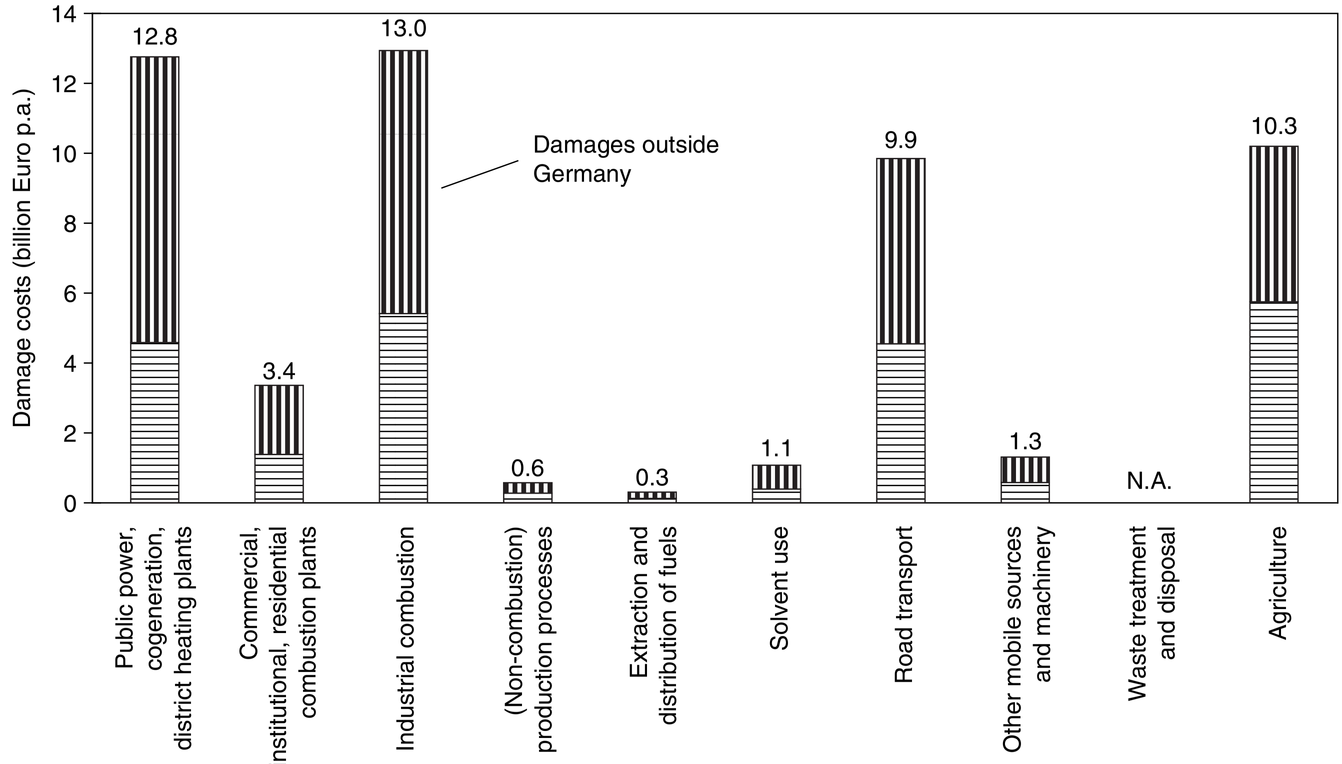

Figure 9.2 Damage costs caused by German economic sectors, broken down into damages inside and outside the country

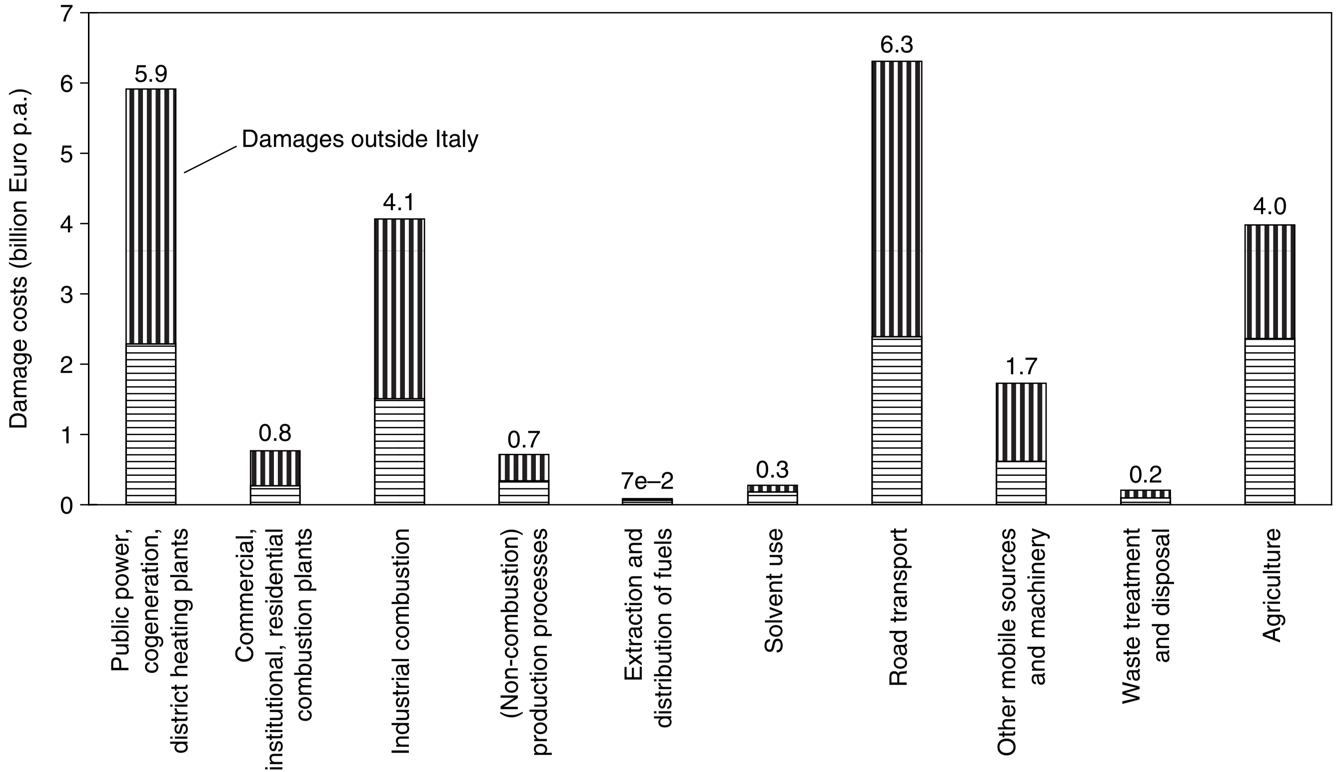

Figure 9.3 Damage costs caused by Italian economic sectors, broken down into damages inside and outside the country

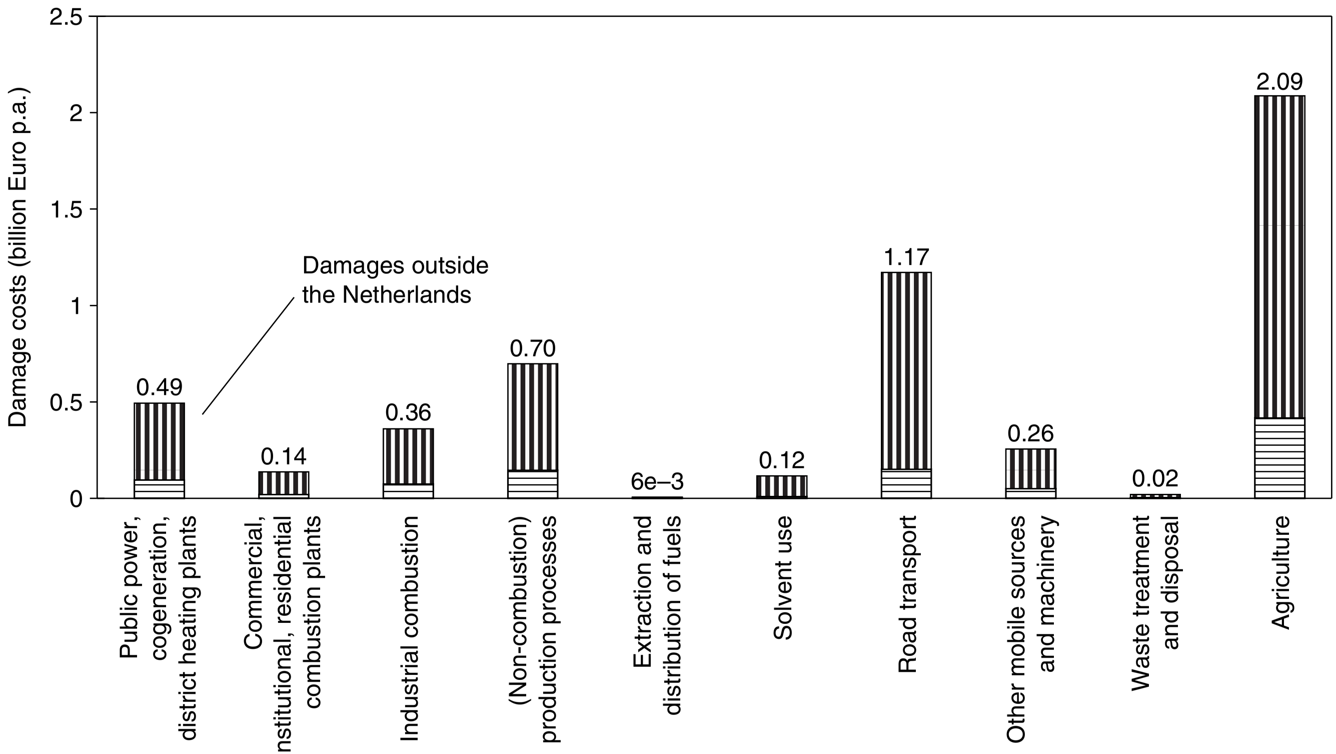

Figure 9.4 *Damage costs caused by the Netherlands' economic sectors, broken down into damages inside and outside the country*

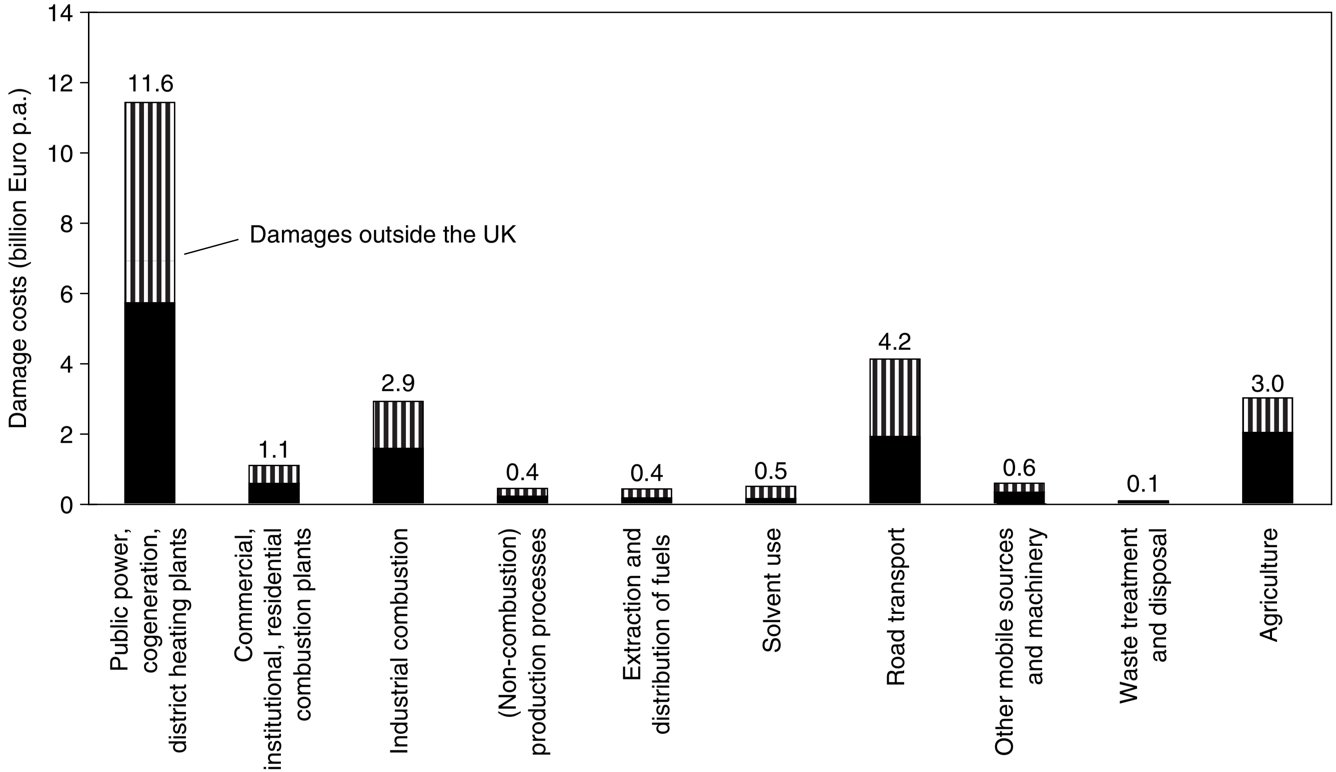

Figure 9.5 Damage costs caused by the UK's economic sectors, broken down into damages inside and outside the country

9.5 ATTRIBUTION OF IMPACTS AND DAMAGE COSTS TO SOURCE COUNTRIES WITHIN THE EUROPEAN UNION

While in the last section the attribution of damages caused by Germany, Italy, the Netherlands and the UK to specific economic sectors was described, in this section the attribution of the damages caused within the EU to source countries is analysed. For this purpose, scenarios including emission reductions by each of the 15 EU member states were applied one after the other to estimate the respective damage distributions. Due to the characteristics of the SROM model, damage costs caused by O_3 concentrations cannot be taken into account in the current analysis (see Section 9.2). So, only SO_2, NO_X and NH_3 emissions could be considered in this part of the study. But it was already seen in the previous section that damages caused by O_3 are small. In the case of German economic sectors only about 1.1 billion Euro p.a. of the 52.5 billion Euro total damage costs (that is, 2 per cent) are due to ozone. So, the analysis will not be much affected by neglecting these damages.

Table 9.23 shows the estimated attributions of damages caused within the EU-15 states. The grey row includes the monetary value of the damages caused within the countries listed in the second row (the 'receptor countries'). The grey columns show the damage costs caused by the source countries. It can be seen that most of the approximately 130 billion Euro p.a. damage costs that occur within the EU are received and caused by Germany, France, the UK and Italy.

'Net imports' of damages are calculated by reducing the damage costs occurring within one country (grey row) by the corresponding damage costs caused by the same country within the EU (first grey column). The country identifiers for net importers are highlighted in the table, while the background for net exporters is darkened. Spain, France, Greece, Ireland, Italy, Luxembourg, Portugal and the UK are identified as net exporters of damages within the European Union.

The white cells in Table 9.23 include the percentages of damage costs caused by the countries of the second column within the countries listed in the second row. Therefore, in each of the columns the sum of the white cells is 100 per cent. The results show that often less than 50 per cent of the damages which occur in the country are caused by its own emissions. This is especially the case for small countries with many EU member states nearby (for example, the Netherlands) while in large countries and countries with fewer EU neighbours most of the damages inside the countries are caused by their own emissions (for example, Germany, Greece).

Table 9.23 Attribution of damage costs within the EU in 1990

Source countries	Receptor countries																
	AT	BE	DE	DK	ES	FI	FR	GR	IE	IT	LU	NL	PT	SE	UK	EU	Non EU
	Damage costs caused by the source countries within the receptor countries (billion Euro p.a)																
EU	2.8	4.5	40.9	2.3	8.9	0.4	21.4	2.0	0.4	15.3	0.1	7.0	1.2	2.1	19.4	128.8	34.9
	Percentage of damage costs caused in the receptor countries (%)															(billion Euro p.a)	
AT	12.2	0.2	0.9	0.6	0.2	0.9	0.2	0.9	0.1	2.2	0.3	0.2	0.0	0.8	0.1	1.2	1.8
BE	1.1	12.3	3.7	3.8	1.2	1.4	3.7	0.0	1.4	0.7	4.8	13.0	0.4	2.6	1.1	4.4	0.4
DE	47.0	14.2	53.8	43.7	4.8	38.7	13.4	2.0	6.9	15.6	33.3	15.6	0.9	49.6	6.7	34.4	17.0
DK	0.6	1.0	0.9	9.2	0.1	4.6	0.5	0.0	0.6	0.1	0.8	1.0	0.0	8.9	0.8	1.2	0.4
ES	1.7	8.6	3.8	1.8	51.8	0.0	16.3	0.0	16.1	5.6	8.9	6.1	50.4	1.0	7.3	13.5	0.4
FI	0.1	0.1	0.1	0.4	0.0	29.5	0.0	0.0	0.1	0.0	0.0	0.1	0.0	1.9	0.1	0.3	0.1
FR	8.7	33.5	15.3	7.2	16.8	2.1	36.2	0.1	11.2	10.3	31.9	23.8	5.9	5.4	11.4	23.2	2.0
GR	1.1	0.0	0.1	0.2	0.1	0.0	0.1	78.0	0.1	2.3	0.1	0.0	0.0	0.2	0.0	2.1	3.7
IE	0.0	0.3	0.2	0.5	0.4	0.1	0.4	0.0	18.8	0.0	0.2	0.4	0.3	0.2	2.1	0.7	0.0
IT	23.3	3.0	6.9	2.5	8.8	1.6	5.9	18.9	2.8	60.1	4.1	2.4	2.6	2.8	1.2	15.8	6.8
LU	0.2	0.3	0.4	0.2	0.1	0.1	0.2	0.0	0.1	0.1	1.6	0.4	0.0	0.1	0.0	0.3	0.0
NL	1.2	8.0	4.6	7.1	1.0	2.3	3.8	0.0	1.4	0.6	5.6	13.8	0.3	5.0	1.8	4.9	0.5
PT	0.0	0.6	0.2	0.1	6.8	0.0	1.1	0.0	2.4	0.1	0.6	0.4	36.0	0.0	0.7	1.6	0.0
SE	0.2	0.3	0.4	1.5	0.0	11.7	0.1	0.0	0.2	0.0	0.2	0.3	0.0	8.1	0.3	0.5	0.3
UK	2.6	17.5	8.6	21.2	8.0	7.0	18.1	0.0	37.8	2.0	7.7	22.4	3.3	13.4	66.5	24.7	1.2

9.6 ANALYSIS OF LINEARITY

The composition of the pollutant mix in the atmosphere which influences the formation of secondary pollutants is determined by emissions of the neighbouring countries as well as the emissions within the country under analysis itself. In this section the sensitivity of the damage estimations towards changes in this chemical environment is investigated.

For this purpose the influence of German emissions on the estimation of damages caused by the Netherlands' emissions was investigated and vice versa. The result was that the estimated damage costs were in each case higher with emissions in the neighbour country than without those emissions. The reason is that the NH_3 concentration is higher without the emissions of the respective neighbour country so that more NH_3 are available to react with the SO_2 and NO_X emissions of the investigated country to form nitrates and sulphates. The influence of German emissions on the Netherlands' damage estimation was about 10 per cent, while the reverse influence was about 1 per cent.

The effects of the inland emissions on the damage estimations are investigated by comparing scenarios with different levels of emission reductions in Germany (see Figure 9.6). It can be seen, for example, that the damage

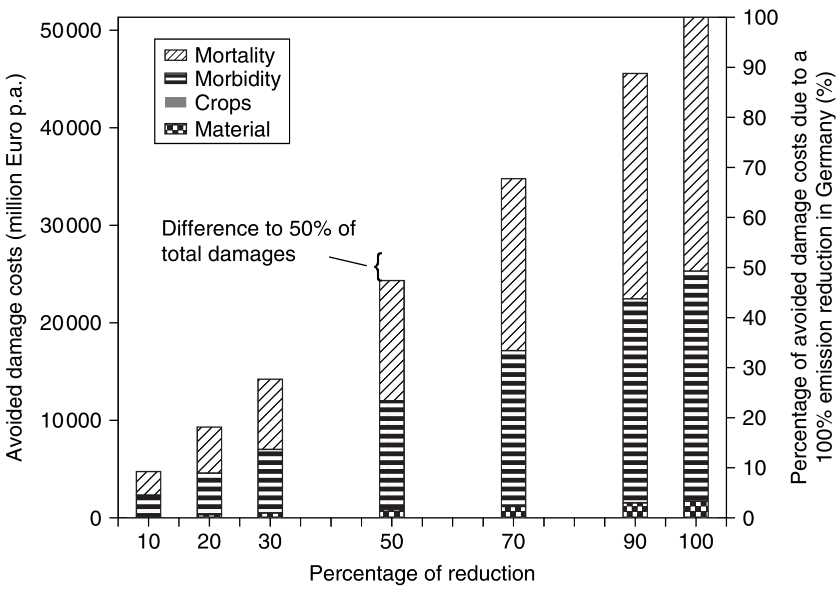

Figure 9.6 Analysis of linearity concerning different reduction stages for German emissions

costs avoided by a 50 per cent reduction of anthropogenic emissions are lower than 50 per cent of the total damage costs resulting from a 100 per cent reduction. The same situation is observed for the remaining reduction scenarios investigated. This means that the effect of the first emission reductions is lower than the effect of the last percentage of emission reduction in Germany. The reason is that less of the SO_2 and NO_X, which are reduced first, can react with NH_3 to form nitrates and sulphates, because most of the NH_3, which also comes from emissions outside Germany, has already reacted. The largest deviation is observed for the damage costs avoided by a 20 per cent emission reduction (9.3 billion Euro p.a.) which are about 10 per cent lower than the 10.3 billion Euro p.a., which represent 20 per cent of the damage costs estimated for the total emission reduction. A similar effect is observed for scenarios evaluated for the Netherlands.

The analysis shows that, as an effect of the non-linearity in the damage estimations caused by influences of the chemical background concentrations, an extrapolation of results derived from a low emission reduction to obtain damage costs avoided by a higher emission reduction is subject to considerable uncertainties. This means that it is also not possible to obtain an exact attribution of damage costs to economic sectors and uncertainties have to be allowed for in the respective impact analysis.

9.7 COMPARISON OF MODELLED AND MEASURED CONCENTRATION DATA

An important part of the damage estimations using the impact pathway approach is air quality modelling. The Source Receptor Ozone Model (SROM) and Windrose Trajectory Model (WTM) were applied in the current analysis (see Chapter 6). Concerning the O_3 model SROM, first comparisons with the EMEP model show a good consistency in the results. Simultaneously, much effort is continuously invested by the developers of the EMEP model to verify their model results by comparing them to concentration data and calculations with former versions of the model (Malik et al., 1996; Simpson et al., 1997). Additionally, the calculations presented in Section 9.4 show that most of the damage costs result from primary and secondary particles, the concentrations of which were computed by applying the WTM model. O_3 damages represent only a small part of the damage costs. Therefore, in the following the focus is on verification of the WTM model.

In order to check the WTM results, modelled and measured concentration data of SO_2 and sulphates for 1990 were compared to each other.

Measured SO_2 concentration data in Germany for 1990 were interpolated to the model grid and compared to the concentration derived with the model. Figure 9.7 shows the ratio of interpolated to modelled SO_2 concentrations. The measured data seem to be overestimated in some regions (ratio >1) and underestimated in some (ratio <1) by the model results. While in most areas this difference is not larger than a factor of two, in some regions of East Germany an underestimation of a factor of six is observed. These are regions with high SO_2 emissions from power plants (see Figure 6.1 in Chapter 6). As the exact location of the measurement stations is not known, the stations taken into account may eventually reflect only the local concentrations near to power stations and therefore may not be representative of average concentrations in the respective 50×50 km^2 grid cell. In this case the interpolation would overestimate the average concentration in this region.

In Figure 9.8 the modelled sulphate concentrations are compared to observed concentration data for measurement stations all over Europe. Extra WTM model runs were performed to calculate the concentration levels at the locations of the measurement stations. The values used were restricted to data measured within the mixing layer assumed for the WTM calculations. The two dotted lines represent an over- and underestimation of a factor of two. More than 85 per cent of the data points lay between these two lines so that most concentration data calculated by the model show smaller deviations to the observed data than a factor of two with a slight overestimation of sulphate concentrations.

The analysis shows that the model seems to underestimate SO_2 in Germany and overestimate sulphate concentrations in Europe. In large parts of the analysed geographical areas the deviation of modelled to measured data is not larger than a factor of two.

9.8 CONCLUSIONS

A methodology was developed to estimate the attribution of damages caused by air pollutants to economic sectors and countries of origin.

The methodology was used to estimate physical impacts and damage costs caused by emissions of the economic sectors in Germany, Italy, the Netherlands and the UK. The damage costs assessed were about 53 billion Euro p.a. for German emissions, 25 billion Euro p.a. for UK emissions, 24 billion Euro p.a. for Italian emissions and 5.2 billion Euro p.a. for the Netherlands' emissions. Expressed as cost per inhabitant of the country that causes the damages, these are 660 Euro per person per year for Germany, 440 Euro per person per year for the UK, 420 Euro per person

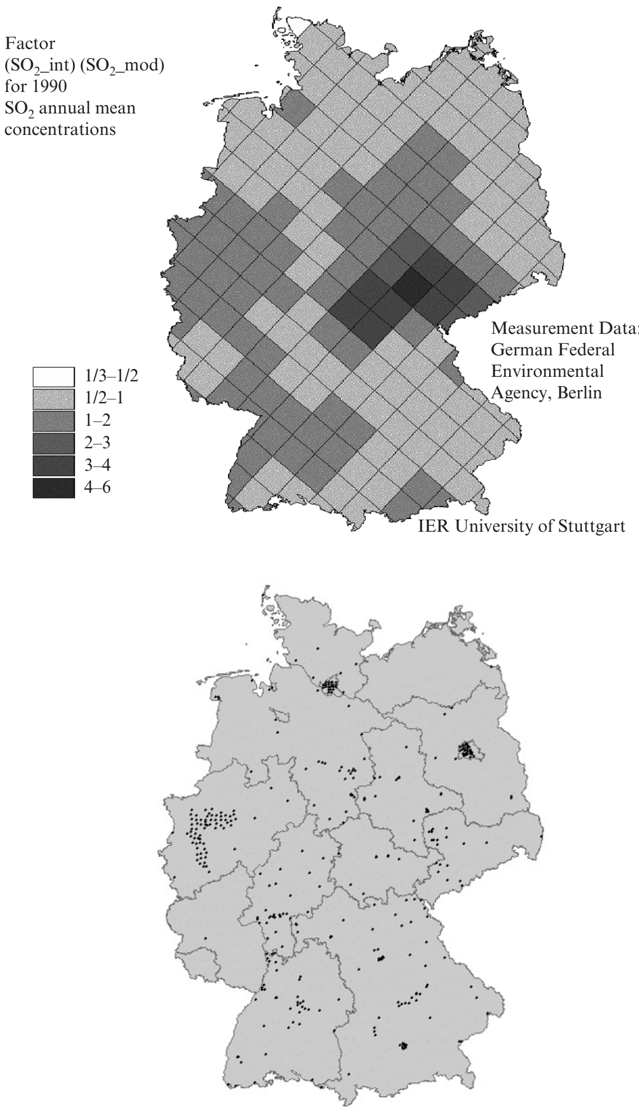

Figure 9.7 Comparison of modelled and measured annual mean concentrations of SO_2 and respective measurement stations for which observed concentration data were reported by the German Federal Environmental Agency, 1990

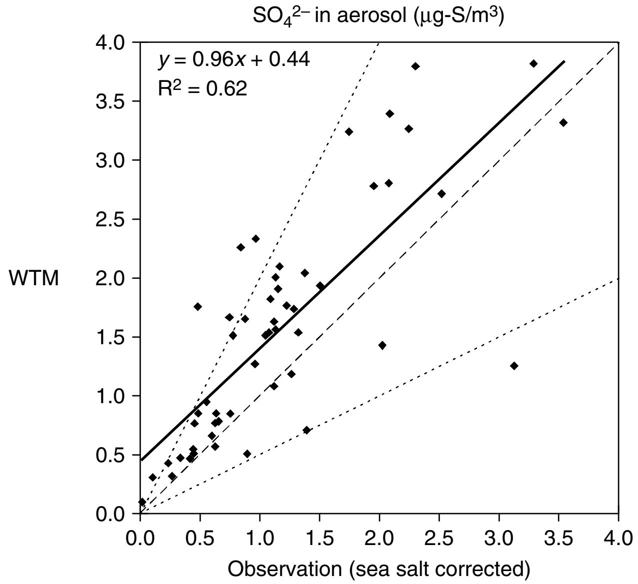

Figure 9.8 Comparison of measured and modelled sulphate concentrations

per year for Italy, and 350 Euro per person per year for the Netherlands. As a percentage of the respective GDP the costs represent 3.5 per cent for Germany, 2.9 per cent for the UK, 2.4 per cent for Italy and 2.0 per cent for the Netherlands. About 88 per cent of the damage costs in 1990 due to German emissions were caused by the economic sectors 'Public power, co-generation and district heating plants', 'Industrial combustion', 'Road transport', and 'Agriculture'. These four economic sectors belong also, for each of the remaining three countries, to the five main contributors to the total damage costs. The results are dominated by the morbidity and mortality effects of the secondary pollutants, nitrates and sulphates, on human health. Rough estimations of impacts due to German emissions of small particles (PM_{10}) for the year 1990 show that these are of the same order of magnitude as the damages caused by nitrates and sulphates. Damage costs caused by O_3 represent only a small part of the total damages for all four countries, for example, about 2 per cent of damage costs caused by German economic emissions.

In addition, the methodology was used to attribute the damages caused within individual EU-15 countries and outside the EU to the source countries within the EU. The results show that damage costs of about 160 billion Euro p.a. were caused by emissions of NO_X, SO_2 and NH_3

resulting from European economic activities in 1990. Seventy-nine per cent of these damages occur inside the EU while 21 per cent occur in non-EU countries. The main contributors to the effects caused within the EU-15 states were Germany (34 billion Euro p.a.), the UK (26 billion Euro p.a.), France (23 billion Euro p.a.), Italy (16 billion Euro p.a.) and Spain (14 billion Euro p.a.). At the same time these were the countries in which most of the damage costs occur within the EU (Germany (41 billion Euro p.a.), France (21 billion Euro p.a.), the UK (19 billion Euro p.a.), Italy (15 billion Euro p.a.), and Spain (8.9 billion Euro p.a.)). It could be seen that the anthropogenic emissions of each of the countries Spain, France, Greece, Ireland, Italy, Luxembourg, Portugal and the UK cause more damages outside the respective country but within the EU than the remaining countries cause inside the country. They were therefore 'net exporters' of damages within the EU in 1990. The largest damage costs inside and outside the EU expressed as a percentage of GDP in 1990 (EC, 1997a) were caused by the countries Greece (7.8 per cent), Luxembourg (3.7 per cent), Germany (3.5 per cent), Spain (3.1 per cent) and the UK (3.0 per cent). The lowest damage costs in terms of the respective GDP were caused by Finland (0.3 per cent) and Sweden (0.4 per cent). The remaining countries cause damages of between about 1.4 per cent and 2.7 per cent of their GDP. Differences to the percentages shown above are caused by damage costs due to O_3, which are not included in the analysis of the damage attribution to EU-15 member states. The most damage per inhabitant was caused by emissions from Luxembourg at 900 Euro per person per year, followed by Germany at 650, Greece at 570, Belgium at 480, the UK at 450, Italy at 400, Austria at 380, Spain and the Netherlands at 360, Denmark at 320, Ireland at 220, Portugal at 160, Sweden at 90 and Finland at 75 Euro per person per year.

The investigations in this study show that the application of Impact Pathway Analysis approach on the attribution of damages to economic sectors and countries of origin represents a major contribution to the development of a Green Accounting System in Europe.

NOTES

1. The damage costs of 8.3 billion Euro p.a. caused by PM_{10} are not included.
2. The damage costs of 2.6 and 2.3 billion Euro p.a. respectively caused by PM_{10} are not included.
3. The damage costs of 3.5 billion Euro p.a. caused by PM_{10} are not included.
4. The damage costs of 8 billion Euro p.a. caused by PM_{10} are not included.

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10. Developments in estimation of damages to crops

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10.1 INTRODUCTION

Air pollution generally has a negative impact on crop yields. In Chapter 8 of this report the monetary value of the impact of ozone on crops in the Netherlands in 1994 was estimated at Euro 150 million. Chapter 8 used the methodology described in Markandya and Pavan (1999) and previously presented in Chapter 1. This methodology uses experimentally derived exposure–response functions that relate ambient concentrations of air pollutants to reductions in yield per unit area to calculate yield reductions. Yield reductions are then multiplied by fixed crop prices to arrive at the monetary damage estimate. We call this the simple multiplication method.

For very small changes in air quality and consequently relatively small changes in yields, this procedure can be defended on the grounds that the changes will not result in any significant changes in prices or other variables that are relevant to the valuation. In the previous phase of the research presented in Markandya and Pavan (1999), however, damages are calculated for current levels of air quality relative to non-anthropocentric background levels. Such changes in air quality cannot be considered marginal. Non-marginal changes in yields will have effects on market prices that may feed back into farmers' supply decisions, choice of crops and cultivators, input use and so on. The simple multiplication approach ignores these economic adjustments. This section examines whether the use of an economic model to simulate market and farmers' responses to changes in air quality will significantly alter the damage estimates derived with the simple multiplication approach. Specifically, this section reports an attempt to produce such an improved estimate for damages to Dutch agriculture. The exercise is confined to damages from two major pollutants: ozone and sulphur dioxide. Damages from ozone are measured as the accumulated exposure to tropospheric ozone over 40 ppbV in the crop-growing season (AOT40c)

(WHO-ECE, 1996). Damages from SO_2 are directly related to its annual ambient concentration in the atmosphere.

Section 10.2 presents a theoretically correct measure of the welfare effects of changes in agricultural yields due to a change in air quality. It outlines the shortcomings of the simple multiplication approach. A number of practical methods that have been used to calculate welfare effects are discussed. In order to illustrate these methods, three studies are described in some detail. Section 10.3 describes briefly the economic model used for the simulation of demand and supply responses. Section 10.4 compares the results of the adjusted damage calculation with the results of the simple multiplication method. Section 10.5 discusses the results and methods.

10.2 VALUING CHANGES IN CROP YIELDS DUE TO AIR QUALITY CHANGES

10.2.1 Theory

The value of changes in air quality on crop yields is equal to its effect on producers' and consumers' surpluses in the markets for these crops (Freeman, 1993). The change in surpluses is caused by the effect of changes in air quality on the marginal productivity of other inputs, which can be represented by shifts of the supply curve. Figure 10.1 illustrates this graphically. Let MC_0 represent the marginal cost curve, or supply curve, of, say, wheat under the current level of air quality and let D represent the (compensated) demand curve for wheat. Equilibrium price and quantity are p_0 and q_0, respectively. An increase in air quality to the non-anthropocentric background level can be represented as a downward shift of the marginal cost curve to MC_1: with the lower concentrations, the same yield as before can be produced with fewer inputs, that is, at lower marginal cost. The marginal productivity of the other inputs has effectively gone up. The changes in consumers' and producers' surplus are as follows. Consumers gain area p_0DAp_1, while the change in producer surplus is p_1AB minus p_0DC – that is, the new producer surplus less the old producer surplus. It cannot be said a priori whether the net effect of the change in producers' surplus will be positive or negative. This depends on the elasticity of the demand curve and the nature of the shift in the marginal cost curve. The more elastic (flatter) the demand curve, the less price will decrease and the more demand will increase. If demand goes up sufficiently to compensate for the reduced price, producers will gain (in terms of Figure 10.1, the 'gain area' ABCF will increase, while the 'loss area' p_0p_1DF will decrease). The net result of all this is a combined gain in producers' and consumers' surplus equal to area

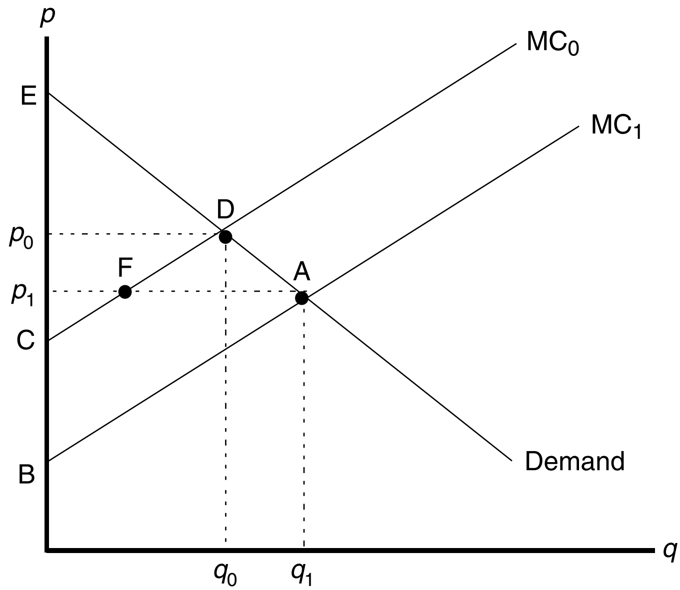

Figure 10.1 The change in producer and consumer surplus

ABCD, that is, the area between the before and after change supply curves, bounded from above by the demand curve.[1]

The analysis illustrated in Figure 10.1 is for one crop only. However, farmers seldom rely on a single crop, and they may respond to changes in the level of air quality by changing their choice of crops or cultivators and the proportions of the total cultivatable area used for each crop. Not all crops are equally sensitive to air pollution. Yields, and consequently profit margins, of different crops will thus be affected to different degrees. Demand has an influence on these changes in profitability too, through the price elasticity of demand, which is not the same for all crops. Analysing such changes in cropping pattern requires a model that completely describes the farmer's cropping choices. The simple multiplication approach ignores potential changes in cropping patterns but empirical studies have suggested that such changes may be important.

Changes in air quality may also result in changes to the relative prices of the inputs and substitutions in aggregate input demand. These changes are caused by changes in the productivity of inputs, caused by changes in air quality and the differential effects of changes in air quality on different crops and regions (Adams and Crocker, 1991). Again, the simple multiplication approach ignores these effects and their impact upon producers' surplus.

The picture becomes even more complicated if one allows for the possibility that the demand curve might shift too. This is not unthinkable, since air quality influences not only production in terms of volume, but can also affect product quality, for example the quality of certain vegetables (for example, spinach) and flowers. In some cases such effects might be treated as simple yield reductions. For example, vegetables that do not meet certain quality standards may be destroyed and never reach the market. In the case that lower quality products do reach the market consumers may be willing to pay less per unit of the product, which can be represented as a downward shift of the demand curve. There have been no empirical investigations of the value of quality changes except for one study by Shortle et al. (1988), concerning soybean quality.

In summary, then, the simple multiplication approach has the following shortcomings: (i) it ignores price changes; (ii) it cannot estimate separate effects on consumers and producers, and ignores the fact that changes in air quality can have differential effects on producers and consumers; (iii) it assumes that areas of crops remain constant and it does not allow for substitution between crops; and (iv) it does not allow for changes in the prices of inputs, nor for changes in aggregate input demand. Moreover, there is the problem of how to treat changes in product quality, but this is not exclusive to the simple multiplication approach but also applies to other methods.

10.2.2 Practical Methodologies

A number of empirical studies carried out during the 1980s, mainly in the United States, estimated the value of changes in agricultural production due to air quality changes. These studies used either mathematical programming models or econometric models based on microeconomic theory. Below both methods are briefly discussed. The discussion draws on Adams and Crocker (1991).

Mathematical programming

Mathematical programming studies approach the problem by using a mathematical representation of the producer's objective and the restrictions she faces in fulfilling this objective. The producer's objective is assumed to be profit maximisation, where profit is a function of prices and quantities of the outputs (crops) and inputs. In realising this objective producers face certain restrictions imposed by the production technology, availability of land and labour, and so on. The parameters of the model can be, but are not necessarily, estimated as part of the exercise. They can, for example, be taken from agronomic studies of crop growth. One of the 'restrictions' the producers face (not necessarily consciously) is the effect of air pollution on yields of the different crops as given by the physical exposure–response relationships. Using the model the optimal (profit maximising) output mix can be calculated for different levels of air quality. Note that these estimated changes are thus not based on relationships estimated from observed behaviour, but are of a somewhat hypothetical character: they represent what would be the optimal response for a profit maximising producer. Models of different 'representative' farms can be used in order to take account of differences in production technology and air quality between different regions.

The model of producers' behaviour can be combined with aggregate demand curves for the crops under consideration. Prices can thus be made endogenous variables, to be determined jointly by forces of supply and demand. With the complete model the outcomes of a change in air quality on prices, output levels of the different crops, levels of inputs and changes in producers' and consumers' surplus can be calculated.

Econometric methods

Econometric models include a number of approaches, which have in common that the relationships used are estimated by econometric techniques. For example, estimated demand and supply curves are combined with exposure–response relationships to estimate changes in production, prices and producers' and consumers' surpluses. Examples of this approach are the Dutch studies by Eerden et al. (1987, 1988) and Bunte et al. (1998), which are discussed in more detail below.

Of particular interest in the category of econometric methods are those based on duality theory. These methods make it possible to estimate the effects of changes in air quality even in the absence of physical exposure–response relationships. In fact, under certain conditions the production function for a crop, including air quality as one of the 'factors of production', can be recovered from the relationships that are estimated from duality relationships.[2]

When using duality theory, the researcher estimates the profit function or the cost function for the producer. A profit function gives the maximum attainable profit as a function of input and output prices, the levels of fixed inputs and any other variable that influences production but is outside the influence of the individual producer, that is, where the individual producer must take them as given. Air quality presents a typical example of such a variable. The general formulation of the profit function is:

$$\pi = \pi\,(p, \boldsymbol{w}, \boldsymbol{x}, q) \tag{10.1}$$

where π denotes profit, p output price, $\boldsymbol{w}$ a vector of variable input prices, $\boldsymbol{x}$ a vector of fixed inputs and q air quality.

The cost function gives the minimum costs necessary to produce some amount of a product as a function of variable input prices, levels of fixed inputs, the particular level of output produced and any other factor that might influence production but is outside the influence of the producer, such as air quality:

$$c = c(\boldsymbol{w}, \boldsymbol{x}, q) \tag{10.2}$$

Both the profit function and the cost function can be derived from the conventional formulation of producer's behaviour as profit maximisation or cost minimisation.

The profit function can be used to directly calculate effects on profits from a change in air quality, by differentiating it with respect to air quality. In the short run, these changes in profits coincide with changes in producers' surplus. In the longer run, farmers can adjust levels of fixed inputs, and changes in profits and producers' surplus will no longer coincide. The profit function can also be used to derive the supply function by equation (10.1) with respect to output price, p. This supply function includes air quality, q, as one of its parameters. The effects of changes in q on supply are therefore easily calculated by differentiating the supply function with respect to q. However, this effect is still calculated on the assumption of constant prices. In order to account for price changes and the secondary effect of these on output, the estimated supply functions will have to be combined with demand functions to calculate price effects. To our knowledge, such an exercise has not been carried out to date.

Data requirements for the dual approach are relatively modest and include data on prices of inputs and outputs, levels of fixed inputs, including air quality, and output levels. Ideally, one would like to have panel data on these variables. With only cross-sectional data there is no variation in the price variables making it impossible to estimate their effect, while time-series data may contain a relatively small variation in pollutant concentrations.[3] A sufficiently large range in the pollutant data is essential, especially if one wants to estimate the effects of considerable changes in air quality.

Advantages and disadvantages of both methods

Mathematical programming techniques have the advantage that not yet realised changes in air quality can be analysed. Econometric methods can also be used to analyse these changes, but, strictly speaking, the estimated relationships are only valid inside the range of available data. Mathematical programming approaches can make use of other sources of information (experimental agronomic research, for example) to specify production technology outside the range of available, observed data. Mathematical programming is of a normative character and assumes perfect knowledge on the part of the producer and a high level of rationality. Dual methods effectively assume this too, but they offer the possibility of actually testing these assumptions. Certain restrictions on the farmer's possible reactions to changes in air quality (restrictions on the cropping pattern, for example) are more easily modelled using mathematical programming. Such, often discontinuous, restrictions are more difficult to analyse using duality techniques. For large changes in air quality that are outside the range of available data, such restrictions might become binding and make mathematical programming a more suitable technique.

10.2.3 Examples

Examples of studies in which damages from air pollution have been studied using these approaches include Eerden et al. (1987, 1988), Bunte et al. (1998), Adams et al. (1982), Howitt et al. (1984), Garcia et al. (1986), McCarl et al. (1989), Adams and McCarl (1985) and Mjelde et al. (1984).

Eerden et al. (1987, 1988) estimated the damage of air pollution to agriculture in the Netherlands. Exposure–response functions for a number of crops were derived from the literature.[4] These were used to calculate yield reductions caused by present-day pollutant concentrations relative to what was considered to be the non-anthropocentric background level, that is, the concentrations that would exist in the absence of any human activities. Using estimated demand functions, output price changes were calculated. The main conclusion from the study was that the economic damages of air pollution fall disproportionately on consumers, while producers see their yield losses (almost) compensated by higher prices. A recent study on the economic impacts of a reduction of crop exposure to NO_x and SO_2 in the Netherlands basically uses the same approach (Bunte et al.,1998). A reduction in ambient concentrations of NO_x of 20 per cent and SO_2 of 60 per cent was estimated to increase producers' surplus by Euro 43 million and consumers' surplus by Euro 41 million. In their assessment, Bunte et al. explicitly assume no adjustments in cropping patterns. They acknowledge that this assumption may not hold in the long run.

One of the first crop damage studies to consider demand and supply responses to changes in air quality is Adams et al. (1982). Adams et al. use price-endogenous mathematical programming to assess the economic damage of ambient ozone exposures to 14 annual crops in Southern California. Howitt et al. (1984) expand the analysis to 38 crops and a wider region. Adams et al. compare the results of their assessment with the results that would have been obtained with the simple multiplication technique. They find that the simple multiplication technique overestimates the damages by 20 per cent, due to its failure to account for mitigative adjustments as well as partially compensating price effects. This failure becomes even more pronounced if one only considers the effect of pollution on producers' surpluses: the simple multiplication technique overestimates producers' surpluses by over 60 per cent (Adams et al., 1982, p. 57). Howitt et al. basically confirm the results of Adams et al. and point out that the percentage changes in total economic surplus are less than the initial yield changes, indicating the mitigating nature of acreage substitution captured in the economic model (Howitt et al., 1984, p. 1126).

Adams and McCarl (1985) use mathematical programming to analyse the impacts of ozone on agriculture in the 'Corn Belt' region of the United States (the states of Illinois, Indiana, Iowa, Missouri and Ohio). This region accounts for over 50 per cent of the United States corn and soybean production and 8 per cent of wheat production. The model used consists of both a micro (producer) level component and a sector component. The micro component consists of linear programming models of 12 'representative farms' in the Corn Belt region. These represent the technical and economic environment of farmers in sub-regions of the Corn Belt. Exposure-response functions describing the effects of changes in air quality on yields are added to the models. The micro models are used to calculate producers' optimal reactions to changes in air quality. The results of these calculations are combined with the sector model in order to calculate aggregate effects. This model contains constant elasticity aggregate demand curves. Aggregate supply effects are calculated by aggregating the calculated micro optima using a special aggregation procedure in order to obtain consistent aggregation; see McCarl et al. (1989). Price effects are calculated by combining the aggregate demand curves and the aggregate supply effects. Price is thus endogenous to the model. Adams and McCarl (1985) and McCarl et al. (1989) show that consumers would benefit from an improvement in air quality (a decrease in ozone concentrations of about 25 per cent), while producers would experience a loss in surplus. The increase in consumers' surplus substantially outweighs the decrease in producers' surplus, resulting in a net increase in total surplus of about 4 per cent ($US 0.668 billion).

The Mjelde et al. (1984) study is an example of an application of duality theory to the estimation of damages from air pollution (see also Garcia et al., 1986). Mjelde et al. estimate a 'translog' profit function for Illinois cash grain farms that includes ozone concentrations in order to estimate the consequences of ozone pollution for farmers' short-run profits (which, in the short run, coincide with changes in producers' surplus). Comparison of the estimates with those obtained from exposure–response studies is hampered by the fact that in the profit function output is a combination of corn, soybeans and wheat while the exposure–response functions are only for one crop at a time. Taking into account the shares of crops in total output, however, the result seems to be reasonably in accordance with those obtained from exposure–response studies.

10.3 THE DUTCH REGIONAL AGRICULTURAL MODEL

10.3.1 Structure of the Model

In this study the Dutch Regional Agricultural Model (DRAM) is used for the analysis of demand and supply responses to air pollution in the Dutch agricultural sector.[5] DRAM takes into account endogenous price effects, substitution of cropping activities and differential impacts on producers and consumers. Furthermore the model combines feed–livestock linkages as proposed by Adams and McCarl (1985) and assumes non-linear total variable cost functions for all model activities as proposed by Howitt et al. (1984). A mathematical representation of the model can be found in Appendix 10.1. More detailed descriptions of the model can be found in Helming (1996, 1997a).

DRAM can be characterised as a price-endogenous, comparative static, spatial equilibrium model. It assumes that farmers act so as to maximise farm profit subject to technical and market restrictions. In DRAM the objective function measures economic surplus (the sum of producers' surplus plus consumers' surplus) under alternative model parameters.

The sector model includes 22 agricultural products marketed outside the agricultural sector and three intermediates (roughage, young stock and manure) produced and consumed inside the sector. The model distinguishes between 14 regions. Seven regions have clayey soil, five have sandy soil and two have peaty soil. Each region is treated like a large, more or less mixed, farm. Data are taken from the Dutch Farm Accountancy Data Network (FADN) to obtain technical and economic parameters per activity, disaggregated to the regional level. Data per activity are connected with the Dutch agricultural census to aggregate to the sector level. The model's base period describes the actual situation with respect to regional and national prices and quantities over the period 1994/95–1996/97.

The model is built around a set of linear regional supply and national demand functions. The national demand functions combine domestic demand and export demand. The demand elasticities used in this report are presented in Table 10.1. Price elasticities of demand for intermediates

Table 10.1 Price elasticities of demand

Ware potatoes	−0.37	Flower bulbs	−1.79
Seed potatoes	−0.79	Beef	−0.69
Pulses	−1.79	Pig meat	−0.69
Onions	−1.79	Poultry meat	−0.54
Vegetables in the open	−1.79	Eggs	−0.49
		Veal	−0.65

Sources: Tonneijck et al. (1998); SPEL-MFSS.

(for example, fodder crops) are determined within the model. Demand for cereals and starch potatoes is assumed to be perfectly elastic, as these crops' prices are regulated by the Common Agricultural Policy (Bunte et al., 1998).

In the arable sector the model distinguishes between 15 marketable outputs: cereals, pulses, sugar beet, fodder beet, ware potatoes, seed potatoes, starch potatoes, onions, other arable products, flower bulbs, four types of vegetables in the open and non-food commodities. It is assumed that these outputs represent total arable production in the Netherlands. Fruit and horticulture under glass are not included in the analysis.

Marketable outputs in the cattle-rearing sector are milk and two types of beef products. Beef represents all outputs from grazing livestock, including male bovines, heifers and suckling cows. In the roughage sector grass, silage and fodder maize is produced. These are used as intermediates in the cattle-rearing sector. In the analysis it is assumed that the gross margins created in the roughage sector are allocated to the cattle sector.

Four marketable outputs (pig meat, poultry meat, eggs and veal) are distinguished in the intensive livestock sector.

In the model's objective function the national profit from agriculture is maximised under the restriction that demand equals supply in every regional market. It is assumed that interactions between regions result from the profit maximising behaviour of the producers, taking advantage of regional price differences (Takayama and Judge, 1971; Labys et al., 1989). Profits are defined as the total gross margin from agriculture.

Commodity balances are included which require national markets to clear. We assume perfectly inelastic supply of purchased inputs. Intermediate balances, describing the interrelationships between regions and activities, consist of roughage, young stock, minerals and manure. Intermediate balances equate regional production, international trade and interregional imports and exports with regional demand for intermediates. (Shadow) prices of land and quotas are derived from restrictions on regional availability.

An important aspect of the model concerns assumptions on production technology. Production technologies are specified as constant proportional (Leontief) production functions. Dairy farming technology is described by a complex linearised system of alternative production methods (Helming, 1997b).

Total variable costs per region and per activity are described by quadratic variable cost functions: the cost of producing a given activity in a region is assumed to increase as the regional level of that activity is expanded (Howitt et al., 1984). The parameters of these activity-based functions are calculated from exogenous prices of purchased inputs per unit (except prices of fertilisation) and shadow values on actual regional activity levels. These shadow values are obtained from an initial model run with activities (areas, livestock number and techniques) restricted to actual, observed figures. This approach is called Positive Mathematical Programming (PMP) and calibrates the model exactly to the actual, observed figures without any loss of flexibility. The PMP approach thus combines some of the advantages of the mathematical programming and econometric approaches discussed earlier. More detailed information on PMP is given by Howitt (1995) and Horner et al. (1992).

10.3.2 Model Application

The model is applied to estimate the effects of two pollutants: Accumulated Ozone Concentrations Above a Threshold of 40 ppbV (abbreviated to AOT40c) and SO_2 (ppb). The average ambient AOT40c values between 1990 and 1995 and the average 1994 ambient SO_2 concentrations in each of the 14 regions defined in the DRAM model are estimated by using the 5×5 km grid data obtained by spatially interpolating regional station measurement data from RIVM (1996), as discussed in Chapter 8. For AOT40c damages a multi-year average was used as there is no clear trend in the levels over recent years (Beck, 1998; Diederen, 1998). The grid-classified data were averaged over the regions by area weighting using the Geographical Information System ArcView 3.0 (1996). The results are presented in Table 10.2.

By applying the exposure–response functions discussed in the methodology chapter (Chapter 4) the yield losses are estimated. The average increase in yields per crop per hectare (ha) on a national scale under scenarios of air quality improvements to the natural background concentrations ($SO_2 = 1$ ppbV and $O_3 = 20$ ppbV thus AOT40c $= 0$ ppbV.h) are presented in Table 10.3. The calculated increases are a function of the assumed sensitivity of crops to air quality, regional O_3 and SO_2 concentrations and regional shares in total crop acreage. Table 10.3 shows that a reduction of O_3 concentrations to natural background levels has much

Table 10.2 The regional average AOT40c between 1990 and 1995 and regional average seasonal seven hour per day mean concentration of SO_2 concentrations in 1994

Region	AOT40c (ppb.h)	SO_2 (ppb)
Northern sea clay area	5 222	1.54
Southern sea clay area	9 377	4.29
River clay area	10 102	2.33
Loss area	10 827	2.78
Northern pasture area	7 081	1.56
Western pasture area	7 516	2.70
Northern sand area	7 041	1.52
Eastern sand area	10 573	1.94
Central sand area	9 188	2.30
Southern sand area	11 056	2.76
Peat colonies	7 221	1.52
Other North Holland	6 719	2.30
Other South Holland	8 309	4.92
Ijsselmeer polders	7 429	1.65

Table 10.3 Changes in yield per hectare due to O_3 and SO_2 reduction to background levels (%)

Crop	O_3	SO_2
Cereals	13.9	−0.8
Pulses	19.3	1.7
Sugar beet	0.0	−0.8
Ware potatoes	18.5	−0.9
Seed potatoes	13.9	−0.5
Starch potatoes	14.7	−0.3
Onions	17.0	−0.9
Other arable products	8.1	−1.0
Grassland	9.4	0.0
Maize	0.0	−0.7
Flower bulbs	0.0	−0.8
Vegetables in the open	8.2	−0.9

Note: A negative number represents a yield reduction.

more impact on yields per crop per hectare than a reduction of SO_2 concentration. In fact the high SO_2 concentrations seem to give rise to increased yields, relative to the yields at the natural background level, except for pulses. We have assumed that the yield increases due to SO_2 pollution are an artefact of the exposure–response functions rather than an observable fact in present-day Dutch agriculture with its high levels of artificial and organic fertilisation. Therefore we have not included the results of the SO_2 scenario in the presentation of the results.

The yield increases due to a reduction of ozone concentration to the natural background level, presented in Table 10.3, are fed into DRAM as exogenous shocks. DRAM calculates a new equilibrium by adjusting production and prices until all markets clear. The numerical difference between DRAM's objective function in the new equilibrium and DRAM's objective function in the 'old' or 'reference' equilibrium is a measure for the economic benefits of ozone control. These economic benefits represent the maximum amount of money that producers and consumers would be willing to pay to reduce ozone pollution and are therefore taken as a measure of the monetary magnitude of current damage.

Because of DRAM's downward sloping demand curves for a number of outputs, the O_3 reduction scenario (which reduces concentrations to non-anthropogenic levels) shows rather dramatic negative price effects for arable crops as compared to the reference scenario (REF, which represents the status quo). The largest negative price effect is for ware potatoes (see Table 10.4). A small negative price effect is calculated for beef and small positive price effects are calculated for intensive livestock products. This is explained by an output reduction in the intensive livestock sector, caused by an increase in the shadow price of manure disposal. This increase in the shadow price of manure disposal results from changes in land allocation and from an increase in manure production by beef cattle.

Tables 10.5 and 10.6 present changes in sectoral production volume and changes in cropping plan under the O_3 scenario. The magnitude of acreage reduction is directly related to the price flexibility of the demand function (that is, the percentage change in price resulting from a one percentage change in quantity demanded) (Howitt et al., 1984). Demand functions for ware potatoes, seed potatoes and roughage crops are characterised by high explicit or implicit[6] price fluctuations. As was also found by Howitt et al. (1984), as a result of the improvement in air quality, production increases, price decreases and acreage decreases. This pattern is not the case for crops with relatively low price fluctuations. The acreage of crops with a relatively low price fluctuation will expand.

Table 10.7 presents the effect of the O_3 scenario on regional and total producer surplus and on consumer surplus.

Table 10.4 Market prices of agricultural products in the reference (REF) and O_3 reduction (O_3) scenarios (Euro per 1000 kg)

Product	Market prices		
	REF	O_3	%
Onion	125.20	111.23	−11.2
Seed potato	220.74	186.77	−18.2
Eggs	712.34	720.03	1.1
Cereals	147.90	147.90	0.0
Pulses	487.93	433.32	−11.2
Other arable	1025.16	1025.16	0.0
Fodder beet	73.28	73.28	0.0
Ware potato	108.74	81.45	−25.1
Starch potato	52.96	52.96	0.0
Sugar beet	52.45	52.45	0.0
Non-food	1372.26	1372.26	0.0
Flower bulb	7857.45	7831.10	−0.3
Vegetable intensive I	622.84	575.88	−7.5
Vegetable intensive II	622.84	575.88	−7.5
Vegetable extensive I	120.16	106.88	−11.1
Vegetable extensive II	271.03	259.09	−4.4
Milk	351.55	351.55	0.0
Beef I	2322.09	2321.69	0.0
Beef II	2764.51	2702.01	−2.3
Poultry meat	725.95	731.58	0.8
Pig meat	1365.70	1375.51	0.7
Veal	2737.45	2741.37	0.1

The O_3 reduction scenario shows an increase in producer surplus from Euro 6054 million in the reference scenario to Euro 6172.73 million, or almost 2 per cent. This increase is mainly allocated to regions with a high share of total land given to roughage crops, starch potatoes and other arable crops. The regions with a high share of land on roughage crops in the reference scenario manage to increase their regional milk production, when roughage supply increases due to air quality improvement. In these regions substitution to arable crops with low price flexibility is less profitable on account of agronomic constraints captured in the supply functions. Farmers in the sandy regions in the Netherlands (the east and south) are negatively affected by the improvement of air quality. In the O_3 reduction scenario consumers gain from lower output prices. Consumer surplus increases by approximately 4 per cent compared to the reference scenario.

Table 10.5 Agricultural output in the REF and O_3 scenarios (1000 tons)

Product	Agricultural output		
	REF	O_3	%
Onion	1071.84	1285.77	19.9
Seed potato	1316.93	1476.99	12.1
Eggs	545.08	542.20	−0.5
Cereals	1556.32	1865.89	19.9
Pulses	15.56	18.68	20.0
Other arable	39.81	47.75	19.9
Fodder beet	129.3	141.97	9.8
Ware potato	3909.17	4271.99	9.3
Starch potato	2778.07	3331.65	19.9
Sugar beet	6664.50	6871.51	3.1
Non food	8.47	9.46	11.7
Flower bulb	49.54	49.84	0.6
Vegetable intensive I	284.57	322.97	13.5
Vegetable intensive II	284.57	322.97	13.5
Vegetable extensive I	883.24	1058.27	27.0
Vegetable extensive II	67.69	70.03	3.4
Milk	11 459.10	11 459.10	0.0
Beef I	160.48	160.5	0.0
Beef II	98.30	99.86	1.6
Poultry meat	791.31	787.98	−0.4
Pig meat	1685.51	1677.18	−0.5
Veal	280.51	280.25	−0.0

Table 10.8 shows the changes in gross margins by sector. Substitution of cropping activities in the O_3 reduction scenario is especially important for the cattle-rearing sector. Producer surplus increases approximately by 3.3 per cent.

Lower prices in the arable sector are compensated by a substitution of cropping activities towards crops with a relatively low price flexibility of demand. Yield increases on grassland and the resulting substitution between maize and grassland on the one hand and also between maize and grassland and arable crops on the other hand, lead to important savings on roughage production crops. The percentage change in the reduction of grassland is less than the percentage increase in grass yields, because of the substitution from maize to grassland. The resulting increase in producer surplus in this sector equals about 2.3 per cent.

Table 10.6 National cropping plans in the REF and O_3 scenarios (ha)

Product	REF	O_3	%
Cereals	199 797.70	210 371.50	5.3
Pulses	5 097.10	5128.20	0.6
Other arable crops	26 951.11	29 874.94	10.9
Seed potato	38 534.65	37 936.65	−1.6
Ware potato	83 319.84	76 765.96	−7.9
Starch potato	62 859.89	65 733.80	4.6
Sugar beet	116 268.60	119 869.60	3.1
Onion	27 606.17	28 283.69	2.5
Flower bulb	18 567.43	18 678.88	0.6
Vegetables intensive I + II	19 241.10	19 704.78	2.4
Vegetables extensive I	25 805.34	26 066.72	1.0
Maize	218 087.40	197 945.80	−9.2
Grassland	1 034 653.00	975 031.20	−5.8
Non-food	22 880.96	29 284.85	28.0
Vegetables extensive II	11 337.17	11 736.14	3.5
Fodder beet	6987.13	7724.08	10.6

Table 10.7 Regional and total producer surplus and consumer surplus in the REF and O_3 scenarios (million Euro)

Region	REF	O_3	Difference (O_3 − REF)
Northern sea clay area	163.17	164.59	1.42
Southern sea clay area	216.13	220.73	4.59
River clay area	156.76	168.19	11.42
Loss area	32.22	32.38	0.16
Northern pasture area	209.51	225.11	15.60
Western pasture area	261.36	277.99	16.63
Northern sand area	245.06	257.06	12.00
Eastern sand area	336.01	330.14	−5.87
Central sand area	145.40	143.32	−2.08
Southern sand area	638.01	633.44	−4.57
Peat colonies	57.52	63.47	5.95
Other North Holland	74.09	74.89	0.80
Other South Holland	18.65	18.96	0.31
Ijsselmeer polders	192.89	190.42	−2.47
Total producer surplus	2746.76	2800.69	53.93
Total consumer surplus	2133.72	2224.51	90.78

Table 10.8 Sectoral income under the O_3 scenario (million Euro)

Sector	REF	O_3	Difference (O_3 – REF)
Cattle farming	1 375.613	1420.327	44.71416
Intensive livestock farming	575.8575	566.7196	–9.14247
Arable	795.2858	813.6479	18.36207
Producer surplus	2 746.76	2800.694	53.93376

10.4 COMPARISON WITH THE SIMPLE MULTIPLICATION METHOD

To mimic the simple multiplication method an SMM-O3 scenario is analysed using the DRAM model. In this scenario, market prices are fixed. Furthermore, it is assumed that all crops produced are sold at these fixed prices, that is, demand is perfectly elastic and there are no supply adjustments. The estimated levels of agricultural output for the reference scenario (REF) and the SMM-O3 reduction scenario are presented in Table 10.9. The benefits of ozone reduction are calculated by multiplying the output changes by fixed market prices (see Table 10.10).

The estimated benefits of O_3 reduction in the SMM-O3 scenario are almost Euro 260 million. In Chapter 8 the damage/benefits of a similar reduction were estimated at Euro 154 million. Of the difference, 80 per cent is due to additional products being taken into account in the DRAM analysis, such as onions and the crops in the vegetable intensive and extensive categories, and only 20 per cent is due to differences in prices used and model features such as changes in inputs for agricultural production.

It is interesting to compare these results with the O3 runs performed with DRAM in the previous paragraph. The results of both methods are summarised in Table 10.11.

The change in producers' surplus in the full model (O3) is smaller than would be expected from the simple multiplication method (SMM-O3). The difference is the result of two opposing forces:

1. The fall of market prices after initial yield increases, which reduces producer surplus as compared to the SMM-O3 scenario. In fact, a model run of DRAM which excluded cropping pattern adjustments in the O_3 reduction scenario showed that the negative price effects due to yield increases would actually make producers worse off. Producer surplus would be reduced by Euro 104 million as compared to the reference situation.

Table 10.9 Agricultural output in REF and the SMM-O3 scenarios (1000 ton)

Product	Output (1 000 tonne)		
	REF	SMM-O3	% Change
Onion	1071.84	1254.16	17.0
Seed potato	1316.93	1500.41	13.9
Eggs	545.08	545.08	0.0
Cereals	1556.32	1772.39	13.9
Pulses	15.56	18.57	19.3
Other arable	39.81	43.04	8.1
Fodder beet	129.30	129.3	0.0
Ware potato	3909.17	4633.57	18.5
Starch potato	2778.07	3185.11	14.7
Sugar beet	6664.50	6664.50	0.0
Non-food	8.47	8.47	0.0
Flower bulb	49.54	49.54	0.0
Vegetable intensive I	284.57	316.30	11.1
Vegetable intensive II	284.57	316.30	11.1
Vegetable extensive I	883.24	1046.81	18.5
Vegetable extensive II	67.69	67.69	0.0
Milk	1 1459.10	1 1459.10	0.0
Beef I	160.48	160.48	0.0
Beef II	98.30	98.30	0.0
Poultry meat	791.31	791.31	0.0
Pig meat	1685.51	1685.51	0.0
Veal	280.51	280.51	0.0

2. Cropping pattern adjustments partly undo the negative price changes. Instead of a loss of Euro 104 million, farmers gain Euro 119 million from ozone control.

Our analysis suggests that the simple multiplication method overestimates producers' gains from ozone control, and hence also the damage to farmers because of ozone pollution. This result confirms the conclusions of the American studies that were discussed in Section 10.2. In a study on ozone pollution in Southern California, Adams et al. (1982) found that the simple multiplication method overestimated producers' surplus by over 60 per cent. Our analysis suggests that the simple multiplication method overestimates producers' surplus by 118 per cent in the case of ozone pollution in the Netherlands in 1994.

Table 10.10 Benefits of O_3 reduction in the SMM-O3 scenario (Euro million)

Product	SMM-O3
Onion	22.83
Seed potato	40.50
Eggs	0.00
Cereals	31.96
Pulses	1.47
Other arable	3.31
Fodder beet	0.00
Ware potato	78.77
Starch potato	21.56
Sugar beet	0.00
Non-food	0.00
Flower bulb	0.00
Vegetable intensive I	19.76
Vegetable intensive II	19.76
Vegetable extensive I	19.66
Vegetable extensive II	0.00
Milk	0.00
Beef I	0.00
Beef II	0.00
Poultry meat	0.00
Pig meat	0.00
Veal	0.00
Total	259.57

Table 10.11 Benefits of O_3 reduction (Euro million), calculated using the simple multiplication method (SMM-O3) and the full model (O3)

	Method	
	O3	SMM-O3
Change in producer surplus	119	259
Change in consumer surplus	200	n.a.
Change in total economic surplus	319	259

The simple multiplication model assumes that prices are fixed. As a result of this, consumers would not benefit from ozone control. However, if market prices were allowed to react to an output increase, consumers would gain from ozone control. The full model calculates an increase of consumer surplus of Euro 200 million.

The total economic surplus (consumer and producer surplus) in our full model is 23 per cent higher than in the simple multiplication method. This finding contrasts with earlier findings. Adams et al. (1982) report a 20 per cent lower surplus because of economic adjustments. In their OECD report on benefit estimation, Pearce and Markandya (1989) cite the result of Adams et al., and conclude that the simple multiplication method provides an upper limit to the economic benefits of a reduction in pollution (Pearce and Markandya, 1989, p. 56). Our analysis refutes this conclusion. Whether the simple multiplication method underestimates or overestimates total economic benefits is a question of the relative magnitudes of the overestimation of producer surplus and the underestimation of consumer surplus. There is no doubt that the simple multiplication method overestimates producer surplus and underestimates consumer surplus, but whether the sum of these two opposite biases will result in underestimation or overestimation of total economic surplus is an empirical question and cannot be known in advance.

10.5 DISCUSSION AND CONCLUSIONS

The ideal approach to measuring economic impacts of air quality improvements in the agricultural sector, assuming that the availability and quality of physical exposure–response functions are not a problem, is a sectoral model characterised by upwardly sloping supply curves and downwardly sloping demand curves for all agricultural inputs and outputs empirically estimated at the farm level and equated at the regional or national level to determine market prices. Until now such a model has not been available and all estimates of economic impacts of improved air quality have been biased as a consequence.

In this analysis the Dutch Regionalized Agricultural Model (DRAM) is used. This model assumes the existence of regional farms. Regional farms compete for the fixed resources to optimise national gross margins from agriculture.

A disadvantage of DRAM is the aggregation bias that is created by grouping individual farms that are not similar into one regional farm and by grouping individual components into the model's activities. Much of the heterogeneity of farms and components is lost (differences in objectives of the farmers, differences in yield per hectare, differences in variable costs per hectare, differences in sensitivity to changes in air quality and so on). The severity of the aggregation bias depends on the magnitude of the above-mentioned differences between farms in a region and components of activities. The second problem is that quality changes are not taken into account. This could be an important problem especially for flower bulbs and vegetables in the open. The third difficulty is the time dimension of the adjustments. Since DRAM is a comparative static model the time-path towards completion of the adjustments is unknown.

Notwithstanding these shortcomings we believe that the damage calculations presented here contribute to existing calculations of agricultural benefits of air quality control. From an economic surplus or welfare point of view, reducing ozone concentration levels is important. However, economic benefits are unequally distributed between activities and regions and hence also between individual producers. The DRAM model estimates a total economic damage of Euro 319 million in 1994 because of ozone pollution to crops. Damages to crops caused by SO_2 seem to be negligible.

The results of the assessment confirm two findings of previous studies. The economic benefits to the agricultural sector of a reduction of ambient ozone levels are likely to be significant and the larger part of these benefits will be passed on to consumers of farm products. Contrary to previous European (Dutch) studies, the study does find a substantial potential for farmers to adjust their activities, especially cropping plans, to changes in ambient air quality. A new, and somewhat surprising result of the present study is the relatively large increase in producer surplus in the cattle farming sector. The mechanism responsible for the linkages between the crop sector and the cattle farming sector is changes in roughage yields and prices. Finally, the study suggests that the regional distribution of farmers' benefits of a reduction of ambient ozone levels is quite skewed and has very little relation with the original regional distribution of ambient ozone levels.

A comparison with the simple multiplication method (yield changes valued at fixed prices) reveals that economic adjustments will result in farmers benefiting less from ozone control and consumers benefiting more. Whether total economic surplus is under- or overestimated by the simple multiplication method is an empirical question. In our analysis the difference in magnitude of total economic surplus as calculated with a full economic model and as calculated with the simple multiplication model is relatively modest. This confirms earlier findings.

NOTES

1. The changes in input demanded might affect the prices of these inputs, leading to changes in surpluses in these markets. If it is thought that such effects are present and might be substantial, a general equilibrium analysis, instead of the partial analysis presented here, would be more appropriate.
2. Although there is no obvious reason why there should be a difference between production functions as used by economists and dose–response relationships as estimated by natural scientists, in practice there is. In the experiments conducted to estimate exposure–response relationships, most of the time only one factor (air quality) is systematically varied while all other factors are held constant. However, production functions give the relationship between level of production and all factors of production simultaneously. An exposure–response function thus gives the relationship between yield and air quality, given some level of all other inputs and is, strictly speaking, only valid for this particular level of the other inputs. To what extent the function is valid for other levels of these inputs depends on the degree of substitution between these other inputs and air quality.
3. However, for a pollutant like ozone this does not have to be a real problem since concentrations of this pollutant exhibit large fluctuations from year to year depending on meteorological conditions.
4. Since this chapter is concerned with the economic methodology of crop loss valuation, the exposure–response functions will not be discussed any further here. (For a discussion of these see Chapter 4.)
5. See Kuik et al. (2000) for a more technical exposition. Due to some differences in assumptions, the numbers in Kuik et al. differ slightly from the numbers presented here.
6. Through shadow prices on roughage balances.

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APPENDIX 10.1

Mathematical Representation of DRAM

Objective function

$$\max Z = \sum_j (Q_j^d(a_j - 0.5b_jQ_j^d)) - \sum_t \sum_r (\kappa_{tr} + \kappa_{tr}^1 X_{tr} + 0.5\kappa_{tr}^2 X_{tr} X_{tr})$$
$$+ \sum_l \sum_r (\nu_l - \zeta_{lr})E_{lr} - \sum_l \sum_r (\tau_l + \zeta_{lr})M_{lr} - \sum_l \sum_r \sum_{r'} \zeta_{lrr'} T_{lrr'}$$

Commodity balances

$$Q_{jr}^d - Q_{jr}^s = Q_{jr}^d - \sum_t \gamma_{jtr} X_{tr} \qquad \textit{all } j, r$$

Intermediate balances

$$N_{lr}^d - N_{lr}^s = \sum_t (\rho_{ltr} - \theta_{ltr})X_{tr}$$
$$-M_{lr} + E_{lr} - \sum_{r'} T_{lr'r} + \sum_{r'} T_{lrr'} \leq 0 \qquad \textit{all } l, r$$

Trade balances

$$\sum_r E_{lr} \leq e_l \qquad \textit{all } l$$

$$\sum_r M_{lr} \leq m_l \qquad \textit{all } l$$

Primary input balances

$$\sum_t \delta_{ktr} X_{tr} \leq B_{kr} \qquad \text{all k, r}$$

Variables

Q_{jr}^d = Demand for output j in region r
Q_{jr}^s = Supply of output j in region r
X_{tr} = Agricultural activity t in region r
N_{lr}^d = Demand for intermediate l in region r
N_{lr}^s = Supply of intermediate l in region r
M_{lr} = Import of intermediate l in region r
E_{lr} = Export of intermediate l in region r
$T_{lr'r}$ = Transport of intermediate l from r' to r
$T_{lrr'}$ = Transport of intermediate l from r to r'

Coefficients

a_j, b_j = Coefficients of the demand function of output j
$\kappa_{tr}, \kappa^1_{tr}, \kappa^2_{tr}$ = PMP coefficients of the quadratic cost function for activity t in region r
ν_l = Unit export price of intermediate l
τ_l = Unit import price of intermediate l
$\zeta_{lrr'}$ = Unit transportation costs of intermediate l from region r to region r'
ζ_{lr} = Unit transportation costs of intermediate l from region r to the national border
γ_{jtr} = Productivity coefficient for output j of activity t in region r
θ_{ltr} = Productivity coefficient for intermediate l of activity t in region r
ρ_{ltr} = Demand coefficient for intermediate l of activity t in region r
e_l = Export demand for intermediate l
m_l = Import demand for intermediate l
B_{kr} = Factor availability k in region r
δ_{ktr} = Factor requirement coefficient k of activity t in region r

11. Forest and ecosystem damages

Ursula Triebswetter and Marialuisa Tamborra

11.1 INTRODUCTION

This chapter will first deal with forest valuation and then with ecosystem and biodiversity assessment. In this book there is an attempt to complete and update the literature mentioned by Markandya and Pavan (1999), with a special focus on both methodological and empirical findings related to biodiversity (Section 11.3).

11.2 FOREST DAMAGE

11.2.1 Causes of Forest Damage

Forest ecosystems are damaged by air pollution either directly or indirectly. The direct effect is caused by air pollution absorbed directly by leaves and the crown, whereas the indirect effect is caused by soil pollution. The literature usually identifies three categories of damage:

- loss of timber production,
- reduced value for recreational activities, and
- reduced existence value.

There is a fourth category of damage, namely ecological damage, which is usually recorded as a *loss in biodiversity*. For forests this should not be considered simply as loss of component tree species but also includes changes in the herbaceous ground cover, having consequent changes in the occurrence and size of animal populations. Valuation of ecological damage is difficult to assess, in part because there is no simple accepted method of quantifying the loss. It is also difficult to ascribe changes in biodiversity to individual processes. Natural systems consist of assemblages of organisms that show complex interactions between each other and with the environment. Processes such as increased levels of pollutants do not always cause immediate change and the 'lag' before an effect is recorded may extend for decades or centuries (Diamond, 1972).

Foresters are now starting to accept that ecological indicators of change can be useful in guiding management (Pyatt, 1971) and in the maintenance of sustainable silvicultural systems (Larsen, 1995). Historically, woodland type was defined simply using the dominant tree species (Tansley, 1935). This creates problems on a European scale as the variation within a tree species across its range (including readily identifiable varieties, sub-species and cultivars) may be greater than that between species. Scots pine (*Pinus sylvestris*), for example, is the most widespread species across Europe, but also amongst the most variable.

The idea of classifying vegetation using all species is not new and phytosociologists have for a long time mapped ecosystems and predicted potential natural vegetation (for example, Neuhauslova and Bohn, 1993). These approaches have been criticised in the past as being subjective and reflecting the bias of their creators, but now more rigorous mathematical approaches are making objective classifications available (Bunce, 1982, 1989).

A further complication is the planting of non-native trees, for example Sitka spruce, Douglas fir, lodgepole pine, eucalyptus and Monterey pine. It is only in the last century that these species have been planted and managed; consequently, as with novel crops, there is no established community of associated species and it is impossible to accurately define ecosystems for them.

Data on forests are regularly measured and collected in Europe within the framework of the Convention on Long-range Transboundary Air Pollution – International Co-operative Programme on Assessment and Monitoring of Air Pollution Effects on Forests (ICP Forests); the results of these surveys are published every year by the European Commission and the United Nations Economic Commission for Europe in *Forest Condition in Europe*. These reports show the results of the Europe-wide forest survey, categorised by country, year, species (by single species and partially grouped into conifers and deciduous), reporting damage classes in terms of defoliation and discoloration.

In a major study (Müller-Edzards, de Vries et al. 1997; Müller-Edzards, Erisman et al. 1997) based on data collected within the framework of the Convention of Long-Range Transboundary Air Pollution – ICP Forests, seven species were selected to be studied. These species are widespread throughout Europe and commercially managed native species. All of them have shown distinct deterioration in forest condition. Critical thresholds in different stress parameters were estimated and tested over a number of plots throughout Europe. For atmospheric pollutants, approximately 20 per cent of the plots showed an exceedance for SO_2 and 90 per cent for ozone. Although 10–25 per cent of the plots showed exceedance for acidity and nitrogen, the effects of those pollutants were predominantly seen as changes in ground vegetation rather than in the measured aspects such as crown condition.

The economic valuation of ecosystems is more difficult than that of a commercial enterprise such as timber producer. However, the value of trees and woodland is now seen to be much greater than simple production, with life-support functions, such as recreation, conservation, landscape protection and carbon sequestration, becoming increasingly important elements.

11.2.2 Physical Damage Indicators

Before presenting monetary valuation of forests, a short assessment of the current physical damage of forests in Germany, Italy, the Netherlands and the UK is given.

According to the German Report on Forest Damage (BMELF, 1996) the state of the *German forests* improved slightly in 1996 compared to previous years. On average over 20 per cent of all trees are significantly damaged (1995: 22 per cent and 1994: 25 per cent). This improvement is mainly due to changes in East Germany and Bavaria. However, the condition of the forest in almost all states in West Germany has worsened (see Table 11.1). At federal level on average almost every other tree is sick. According to a study undertaken by the German Federal Environmental Agency, which looked at about 10000 national and international research reports, the essential cause-and-effect relationships concerning forest damage can now be identified (*Süddeutsche Zeitung*, 1997). It is now proven that forest damage is caused by anthropogenic air pollution. There is a direct impact pathway through crown and leaves and also an indirect damage pathway through soil. Although the catastrophic scenarios, of which some researches were afraid in the early 1980s, have not become true, the forest ecosystems in Germany continue to be seriously endangered.

Oak trees are suffering from rapidly increasing damage: in 1996 about 48 per cent of them showed signs of damage (1991: 31 per cent). Spruce and fir are doing better, whereas beech trees are still severely damaged. The reason for the general improvement is assumed to be found in the cool and humid summer of 1996 and in the collapse of heavy industry in East Germany. In fact, there are fewer SO_2 emissions because less coal is burnt and large combustion installations have implemented desulphurisation measures. However, SO_2 still plays an important role in Saxonia because of emissions coming from the Czech Republic.The danger of soil acidification has again increased, since compensation effects have decreased. This is due to the fact that some pollutants that compensate each other, e.g. acid

Table 11.1 Forest damage in the federal states of Germany

Federal state	% of trees without visible damage in 1994	% of trees without visible damage in 1996
East Germany	37	48
Mecklenburg-Vorpommern	41	56
Saxony	40	52
Saxony-Anhalt	35	57
Thuringia	22	27
Brandenburg	42	48
West Germany	n.a.	n.a.
Bavaria	31	47
Baden-Württemberg	35	25
Berlin	32	37
Hessen	25	26
Rhineland-Palatinate	39	36
Saarland	53	47
Lower Saxony	42	48
Schleswig-Holstein	50	43
North Rhine-Westfalia	49	n.a.
Hamburg	52	50
Bremen	55	58
Germany	36	43

Source: BMELF (1994 and 1996).

(which contains SO_2) and particulate matter containing limestone, have nearly disappeared, as is the case for particulate matter and as a consequence there are no more compensation effects for acid (Wienhaus, 1996). Although SO_2 concentrations have been continuously decreasing in New Hampshire for nearly 30 years, plants in the selected forest area have not grown since 1987. The reason is that acid rain has washed out calcium, magnesium and natrium ions.

The German Ministry of Research defines trees to be healthy when they have not lost more than 45 per cent of their needles or leaves. Although the condition of the crown of a tree is a good measure to compare and classify damage throughout Europe, it is, according to forest experts, also necessary to look at the roots of trees. In order to identify the causes of disease the US research results confirm that chemical values, soil condition, biotic and meteorological figures must be included. Only then can the state of the crown be used for diagnosis.

Table 11.2 Coefficient of forest coverage in the Italian regions

Region	Coefficient (%)
Piemonte	29.3
Valle d'Aosta	25.9
Lombardia	25.1
Bolzano – autonomous province	42.6
Trento – autonomous province	57.9
Veneto	19.1
Friuli Venezia Giulia	36.9
Liguria	69.1
Emilia Romagna	20.5
Toscana	42.7
Umbria	39.8
Marche	23.1
Lazio	27.1
Abruzzo	29.8
Molise	29.2
Campania	27.9
Puglia	7.7
Basilicata	29.5
Calabria	38.3
Sicilia	10.4
Sardegna	40.5

Source: Ministry of Environment (1997).

In order to limit forest damage a broad variety of trees should be planted instead of monocultures. It is also important that trees are exposed to as few pollutants as possible, that is, the borders of forest areas need special care. Moreover, calcium or magnesium containing particulate or fertilisers should be used in order to reduce the degree of soil acidity.

Major sources of pollution are still NO_x and carbon oxides from traffic, ammonium from agriculture and SO_2 from power plants. In East Germany industrial emissions are still an important source of pollution.

Before assessing forest condition *in Italy*, it is important to give some data on forest extension in Italy. Forest coverage in Italy has increased since 1970. In fact, in 1970 forests covered 5285 hectares whereas now they cover 5615 hectares. This represents 30 per cent of the Italian territory. A breakdown by region is given in Table 11.2.

Recently, Italian institutions have been encouraging the introduction of environmentally sustainable practices in forest management, which take into account hydrogeological stability and environmental requirements, as well as economic and social issues. The paper industry is also moving towards a more responsible use of resources, encouraging environmental certification and recycling.

At European level, Regulations No. 3528 of 17/11/86 and No. 1696 of 10/6/87 define common criteria for the measurement, collection and transmission to the European Commission of data about forest conditions. According to these regulations, samples need to be collected every year on a grid with a spatial resolution of 16×16 km^2 in all member states. Two different parameters have to be evaluated: tree defoliation and discoloration: for each tree, a percentage of defoliation and discoloration has to be evaluated, compared with a healthy tree. Depending on this percentage, samples are divided into classes, following the structure shown in Table 11.3. These criteria are also applied in carrying out the survey for the Convention on Long-Range Transboundary Air Pollution – ICP Forests. For the purpose of the 1995 survey, 4549 trees in 210 plots were assessed in Italy. Approximately 56 per cent of all trees were classified as defoliation class 0 and 15 per cent as defoliation class 1 whilst 16 per cent were moderately (class 2) and 3 per cent severely defoliated (class 3). Trees are considered damaged when they fall into classes 2 to 3. No clear trend was observed for Italy. However, the rate of tree damage in Italy has always recorded a rate below 20 per cent: 18.9 per cent in 1995, 19.5 per cent in 1994 and 17.6 per cent in 1993.

In Italy, the survey is conducted on a finer grid resolution (3×3 km^2) by

Table 11.3 Definition of defoliation and discoloration classes for forest condition characterisation

Class	Degree of defoliation/ discoloration	Percentage of leaf loss/discoloration	Percentage damage in Italy*
0	None	0–10	62
1	Slight	11–25	17
2	Moderate	26–60	18
3	Severe	>60	3
4	Dead		0

Note: * Figures rounded up to 100%.

Source: Classification according to Regulations no. 3528 of 17/11/86 and no. 1696 of 10/6/87; UNECE and European Commission (1996).

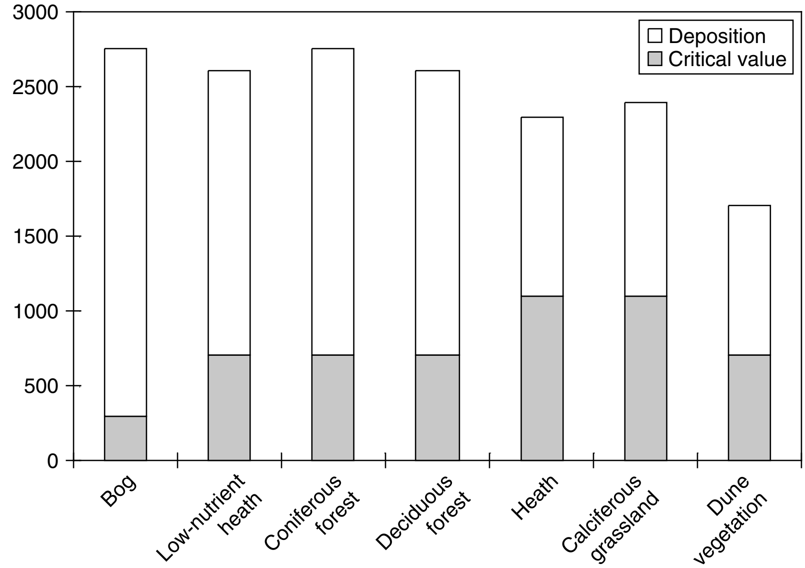
the Ministry for Agricultural Policies, and results are then transferred onto the coarser European grid (16×16 km^2). Some results from this survey have been published by ISTAT in *Environmental Statistics* (ISTAT, 1996), regarding the condition of Italian forests for the years 1990–1994 (data for 1994 are also categorised by species and region). It is not clearly indicated whether these results refer to discoloration or defoliation data, but they seem to be a synthesis of the two.

Italian data on forests are collected by the Ministry for Agricultural Policies, reporting the following:

1. for each Italian region, the percentage of damaged trees (regardless of whether discoloration or defoliation is present, and without damage class specification) – years 1988–94;
2. for each tree species, the percentage of damaged trees (regardless of whether discoloration or defoliation is present, and without damage class specification) – years 1988–94;
3. for each tree species/region, the percentage of damaged trees (regardless of whether discoloration or defoliation is present, and without damage class specification) – years 1988–92;
4. the percentage of Italian trees falling into each defoliation class, for all species and divided by deciduous and conifers – years 1989–96; for year 1997, the division in 10 per cent defoliation classes is also reported.

The Netherlands contain about 470 000 ha of natural area, including forests (71.5 per cent), heath (7 per cent), nutrient-poor grasslands (7 per cent), dunes (5 per cent), marshes (4.5 per cent), tidal marshes (2.5 per cent), bogs (2 per cent) and sand shifts (0.5 per cent). Forests and natural areas occupy about 13.3 per cent of the total land area of the Netherlands. In the period 1983–93 the size of natural areas has increased somewhat in the Netherlands. But this has coincided with a decrease in the quality of these areas (for example, loss of biodiversity). The current pattern of nature and forest areas is highly fragmented. About 80 per cent of natural areas are smaller than 10 ha; only 1 per cent are larger than 500 ha (RIVM et al., 1997).

The forest area in the Netherlands has increased by 32 per cent since 1990 due to afforestation, although the health of the forest ecosystem is not very good. The health of forests is measured in terms of 'vitality'. A vital forest is one for which a positive future is not in doubt. Less vital forests could develop positively, but there are signs that developments could become negative. The future of forests of little vitality is bleak unless there is a rapid change in the factors that determine their health. Non-vital forests have no future, unless there are drastic and immediate changes in the health-determining factors. The vitality of Dutch forests has remained



Source: RIVM (1997).

Figure 11.1 Average nitrogen deposition levels and critical values for a number of biotopes in the Netherlands, 1995 (in mol/ha/year)

roughly constant over the past decade: about 25 per cent is barely or non-vital. There are changes for single tree species, however. The vitality of the Douglas, the fine spruce and the Corsican pine are declining significantly; only the vitality of the Scots pine is improving (RIVM 1997). Critical loads of nitrogen deposition are exceeded almost everywhere. Fig. 11.1 shows the average nitrogen deposition levels and critical loads for a number of biotopes in 1995. The situation is worst for bogs, while the situation is relatively better for dune vegetation.

In the *United Kingdom* (UK), the Forestry Commission have been monitoring the condition of forests since 1984, producing annual reports (for example, Innes and Boswell, 1990). Approximately 50 plots distributed across the UK are used for each of five different tree species (Sitka spruce, Norway spruce, Scots pine, oak and beech). Monitoring involves an assessment of crown density (recorded following Council Regulation (EEC) No. 2995/89) along with 44 other characteristics (some species specific). Correlations between the different characteristics for each species have been calculated. The difficulty is in attempting to relate forest decline to causal factors. Occasionally, for some species, the effects can be attributed to a single factor, for example, the decline of the elm in the UK in the 1970s was the result of an aggressive strain of the fungus *Ophiostoma ulmi* (Buisim.) which moved in from North America. Although other pathogens can be identified, atmospheric pollution may still contribute by altering the condition and susceptibility of individual trees to attack.

The conclusion on changes in forest conditions in the UK between 1989 and 1992 was that drought is the major factor affecting the condition of trees. There are a number of factors which can support this observation; first, the UK is a maritime nation and consequently sees greater variation in rainfall between years; second, the species planted for commercial purposes are predominantly non-native and so have not evolved the same tolerance of local climatic fluctuations seen in native species. For some species, pollution effects have been tentatively identified, namely ozone contributing to dieback in beech and limiting the extent of flowering in Scots pine and SO_2 being related to the reduction in crown density in Norway spruce (Mather et al., 1995). It may be significant that the species seen to be stressed by atmospheric pollutants are native.

Johnson and Taylor (1989) identified three main routes for atmospheric pollution damage to trees: soil acidification, gaseous effects and nitrogen deposition. The same mechanisms will operate across the whole ecosystem. There is evidence of the mechanisms affecting woodland ecosystems, for example the species richness of ground flora in oak woodland is closely related to the pH of the soil, which in turn is influenced by acid rain (Brunet et al., 1997a and b).

However, whilst the main floristic gradient in Swedish oak and beech woodlands is related to soil acidity (Brunet et al., 1996), it has not been possible to detect changes in ground flora in the direction of a more acid-tolerant group of species in a 10-year study (Brunet et al., 1997a and b).

11.2.3 Direct Commercial Value of Forests

The direct commercial value of forests is limited, but there is high variability between countries depending on their natural endowment. In *Germany* in 1995 timber revenues exceeded Euro 2500 million, whereas timber revenues from *Dutch forests* have been estimated to be about Euro 25 million per year. The total production value of ‘forestry products’ in the Netherlands was Dfl 399 million (Euro 180 million) in 1995 (CBS, 1998). This includes not only timber but also Christmas trees and ornamental trees. The production of wood and wood products in volume terms was about 1000 million m^3 in 1995. Revenues from hunting do not exceed Euro 5 million per year in the Netherlands. Thus, the direct commercial value of forests and ecosystems is limited. Moreover, the total

Table 11.4 Roundwood production in 1993, 1000 m^3 underbark

Country	Production
Germany	36 156
Italy	9860
The Netherlands	1403
United Kingdom	6195

Source: Eurostat (1996).

economic value of forests and ecosystems in the Netherlands is unknown. The main economic value of nature in the Netherlands lies in its recreation and tourism function, its ecological functions and in its existence value.

Data on commercial value from forests are not easily available in *Italy*. Input–output tables existed at the time of our project only for the year 1988.

Only around 10 per cent of the *United Kingdom* is covered by woodland, an area of 2.4 million hectares, which is well below the European average of 25 per cent. Of that total, 2.2 million hectares are being productively managed by the Forestry Commission (FC) or supported by government grants. There is a government policy commitment to increase woodland area with targets of a doubling of the area in England, and a 50 per cent increase in Wales and Scotland. Forest area in the UK has more than doubled since the start of the twentieth century. The FC estimates its timber production as 4.7 million m^3 for 1996–97, most of which is softwood (Statistics, 1998). This is about 30 per cent less than in 1993 (see Table 11.4 for a comparison of roundwood production in the countries under consideration in this study).

11.2.4 Selected Studies on the Economic Valuation of Forests and Parks

A large number of studies have assessed the value of components of nature in specific contexts. Several assessment methods have been used, including the Travel Cost Method (TCM), Hedonic Pricing Method (HPM) and the Contingent Valuation Method (CVM) for calculating the value of specific forests and parks in Europe. A number of studies have used values from other studies and contexts (using Benefit Transfer techniques). However, at this stage it is impossible to provide a single figure for each country, as studies are sparse and values cannot simply be aggregated. There are some overall studies aimed at calculating the willingness-to-pay for preserving nature in a country, but their reliability is relatively low. These studies, however, are aimed at valuing forests, especially in relation to their recreational function and not at valuing their damage. Clearly, people′s willingness-to-pay also reflects forest conditions and is inversely correlated to forest damage, but these have not been explicitly quantified.

As an example, an overview list of some relevant studies with their authors is provided in Table 11.5, based on Hampicke (1996). It is significant that the results obtained using different approaches are similar despite the fundamental methodological differences (Hampicke, 1996). There are noticeable differences in the willingness-to-pay in different countries which may be partly explained by income differences (Hampicke, 1996). In the following a more comprehensive view is given for Germany and Italy.

In *Germany* Ewers et al. (1986) and Joebstl (1989) were among the first researchers to try to monetise forest damages. Since the early 1990s, efforts to put a monetary value on the recreational value of forests have been undertaken relatively often. Some of the most recent German CVM studies for estimating the recreational value of forests are limited to particular forest areas and have been undertaken by Löwenstein (1994), Bergen and Löwenstein (1995), Luttmann and Schröder (1995), Oesten and Roeder 1995, Dunkel et al. (1994), Elsasser (1996) and Klein (1994).

Due to site-specific differences, however, the German studies cannot be aggregated to obtain an overall estimation for all German forest areas. Still, the general usefulness of the methodology is clear. Some of the German studies mentioned are discussed in further detail below.

One of the most elaborate studies has been undertaken by Elsasser (1996), who implemented both a CVM and a travel cost approach for forest areas in Hamburg and the Pfälzerwald. In the CVM, daytrip visitors in both Hamburg and the Pfälzerwald as well as holiday guests in the Pfälzerwald were asked for their willingness-to-pay: daytrip visitors in Hamburg gave a hypothetical annual entrance fee and holiday guests in the Pfälzerwald elicited a value for a 'forest tax' which they would be willing to pay for the entire duration of their stay. The presentation of the data is based on weighted averages.

In both areas daytrip visitors showed an average willingness-to-pay of about DM 100, which visitors were willing to pay for an annual 'entrance ticket' to the forest. One potential bias on the results was tested by giving the visitors the opportunity to revise their initial answers during a connecting question. It is argued that without such an opportunity answers would be underestimated. People who revised their answers on average doubled their WTP during the connecting question. Without the connecting question WTP would have been only two-thirds of the WTP eventually stated in Hamburg, and for the Pfälzerwald only 58 per cent.

However, WTPs in different parts of the analysed forests can differ significantly. Ad hoc aggregation of different sites risks concealing actual

Table 11.5 Survey of international studies on the willingness-to-pay for recreation in forests

Authors	Method	Results
Bojö, 1985	TCM CVM	US$1.92 per hectare and year US$230 000 per year for total forest area under examination; US$4.50 per person and visit
Nohl and Richter, 1986	value of time	DM 4.87 per hour; DM 13.4 billion for West Germany in 1986
Hanley, 1989	TCM CVM	£1.70 per visit £1.25 per visit as maximum acceptable entrance fee
Willis and Benson, 1989	TCM	£1.26–2.51 per visit; £12.37–231.51 per hectare and year
Willis, 1991	TCM	£1.34–3.31 per visit
Willis and Garrod, 1991	TCM	£1.43–2.60 per visit (zonal method b) £0.40–2.32 per visit (individual method) £0.06–0.96 per visit (individual method, different statistical processing)
	CVM	£ 0.43–0.72 per visit
Whiteman, 1991	TCM	£1.52–2.83 per visit, £200 per hectare and year
Karameris, 1991	TCM	DM 4.10 per hour
Merlo and Signorello, 1991	TCM CVM	Lire 5490 per visit Lire 2397 per visit
Bishop, 1992	CVM	£0.41–1.34 per visit as maximum acceptable entrance fee; £2.09–1.45 per hectare and year
Nielsen, 1992	CVM	SFr 4400 per person living in the region
Löwenstein, 1994, Bergen and Löwenstein, 1995	TCM CVM	DM 2.28–8.77 per day of visit DM 4.56 per day of visit
Luttmann and Schröder, 1995	TCM CVM	DM 5.17 per person (incl. overnight stay) DM 2.59 per person (incl. overnight stay); 61% of forest area in area under examination
Oesten and Roeder, 1995, according to Dunkel et al., 1994 and Elsasser, 1996	CVM	DM 114.07 for annual entrance fee in Hamburg DM 100.81 for annual entrance fee in the Pfälzerwald

Source: Hampicke (1996).

Table 11.6 Willingness-to-pay of daytrip visitors (in DM)

Area	n	Average	Average, 5% trimmed	Median	Min.	Max.	VC
Hamburg	1028	114.07	92.47	75	0	5200	129.2
Pfälzerwald	1127	100.81	80.26	60	0	3000	164.9

Source: Elsasser (1996).

regional differences. Regression analysis of the WTP showed that *ceteris paribus* the type and site of forest has an impact on the evaluation of its recreational function (for example, WTP was about one-third higher in forests close to inner city areas).

Concerning the WTP of holiday guests it turned out that tourists were on average willing to pay about 30 DM for the entire stay, that is, about 8 DM per day of visit. Again, if a connecting question had not been suggested, the WTP obtained would have been an underestimate. Regression analysis showed that the values obtained were consistent with theoretical expectations. Only the expected negative influence of travel distance on WTP could not be proven. Elderly people revealed a relatively lower WTP. According to the hypotheses about non-respondents constructed with the help of regression analysis average WTP would need to be reduced by 10 per cent (Elsasser, 1996).

The average WTP for the total population of Hamburg was about 120 DM in the forest areas analysed. This WTP is weighted by the frequency of forest visits.

Within all limits of comparability it is worth noting that a strategic experiment done by Klein in 1994 arrived at similar results to Elsasser's study. Klein examined the strategic response behaviour of two different groups of forest visitors. He developed two questionnaires, of which the first created the incentive to exaggerate the interviewee's willingness-to-pay and the second offered the incentive to understate one's willingness-to-pay for the recreational value of the forest. The result of this survey among 70 daytrip visitors in the Haardt forest on the northern edge of the Ruhr area was a willingness-to-pay of about 100 DM per person per year. This mean value corresponds with Elsasser's results.

Another German study which tried to quantify the recreational value of forests was undertaken in 1994 by Löwenstein. He used both the travel cost method and the contingent valuation method for a comparative analysis of forest recreational value in the Südharz. The CVM results for holiday guests per day are between DM 2.83 and 3.83 (depending on the use of zero responses) and vary between DM 52 and 70 for the entire stay. These results are plausible in comparison to the Elsasser study because the average stay of holiday guests in the Südharz is longer than in Elsasser's study. This may explain why it is that the average WTP for the entire stay is higher and that for each day smaller than in Elsasser's research.

Beyond the statistical problems shown extensively in the literature one specific objection could be raised against the results presented (Hampicke, 1996): the participants of the study revealed a gain in consumer rent according to the specific biotope visited. Theoretically, the consumer rent could be taken away as an entrance fee. However, this does not mean that the total willingness-to-pay is oriented towards the forest-specific characteristic of a biotope. People who just want to go for a walk, irrespective of whether in a meadow or in a forest, do not express a forest-specific willingness-to-pay. Generally, one would need plausibility arguments in order to check whether this objection applies. For example, around Hamburg (Elsasser, 1996), where forests are a scarce resource, it may be assumed that visitors consciously chose the forest for walking. In regions with more forest resources this cannot be assumed with certainty.

Since none of the German studies reported above consider how environmental damage impinged upon the recreational value of the forests, an attempt is made to look at the environmental data for each site retrospectively and see if a link can be made to the monetisation of environmental damage. The studies looked at four forest areas:

- the Hamburger Wald (Elsasser, 1996),
- the Pfälzerwald (Elsasser, 1996),
- the Haardt (Klein, 1994) and
- the Südharz (Löwenstein, 1994).

The city of Hamburg owns a forest area of 3200 ha (5 per cent of the total city area). Six large forest areas, mostly some 10–20 km away from the city centre, account for the main forest area. One half of the forest is coniferous, the other deciduous. In areas with sandy soil types coniferous trees dominate, whereas in other areas there are more beech and oak trees. One of the six forest areas is a nature reserve. The Pfälzerwald is a nature park and one of the biggest forest areas in Germany (135 000 ha). About 75 per cent of the nature park is covered by forest, which consists of 50 per cent pines, 20 per cent spruce, 20 per cent beech trees and 10 per cent oak trees. There are a lot of hiking opportunities and cultural attractions which make the Pfälzerwald an interesting holiday resort. The Haardt is the eastern part of the Pfälzerwald and is as large as 5500 ha. It is a mountain chain of 9 km in length. This area is less natural than the western or northern part of the Pfälzerwald, but offers more hiking and horse riding routes for visitors and is very popular for weekend excursions. The Südharz is a unique resort area with a variety of recreational opportunities such as hiking, walking or skiing. There is also an infrastructure for spa activities.

A ranking of the forest areas could be done in terms of their environmental importance. The Pfälzerwald and the Haardt as a nature park would rank first, followed by the Hamburger Wald as a nature reserve (at least in part) and then finally, the Südharz. However, this classification does not take account of damage classifications. Since it is impossible to obtain accurate data on forest conditions from the forest administration, a rough classification of average damage on a state level can be found in Table 11.1 of this chapter.

According to this inventory the forest of Hamburg was least damaged with 52 per cent of trees without visible damage, followed by the Südharz with 47 per cent[1] and Rhineland-Palatinate (Pfälzerwald) with 39 per cent. However, the differences in forest conditions are not reflected in varying values for the recreational value. It could well be the case that either forest visitors do not realise differences or that the study designs (which originally did not intend to measure the impact this study is interested in) are just not capable of giving an answer to the question of how much forest damage lowers recreational value. These considerations were also confirmed in contacts with the Institute for Forest Economy, University of Freiburg.[2] One of the major problems in this context is the lack of a deeper understanding of what forest visitors perceive and, second, how and/or whether actual perception influences people's willingness-to-pay. A rather drastic example which could give rise to the assumption that recreational value is not influenced by forest damage is the phenomenon of 'forest damage tourism' in Germany. For example, forest visitors in the severely damaged forests of the Erzgebirge seem to enjoy their stay there very much.

Moreover, it is methodologically very difficult not to influence people's perceptions in CVM analysis. Biases in favour of socially positive behaviour must be treated very carefully.

In *Italy* many empirical studies have been carried out on the valuation of forests, using the Travel Cost Method, the Hedonic Pricing and the Contingent Valuation Methods. The first study appeared in the mid-1980s. However, except for a very small number of studies, all aim at evaluating the use value that are captured more easily, rather than the total economic value. No comprehensive evaluation of the Italian forests has yet been undertaken. In fact, these studies refer to specific locations, such as national and regional parks. Because they are site-specific, they cannot be aggregated into an overall estimation. However, we provide a synthetic table (Table 11.7) showing for each site the methods used and the results obtained in terms of monetary valuation.

Table 11.7 Summary of Italian empirical studies on environmental benefit valuation of forests

Authors	Site	Method	Results (in current Euro)
Merlo (1982)	Val Rosandra (TS), forest recreation	TCM	0.875–3 (summer)
Boatto et al. (1983)	Tarvisio National Forest (UD), forest recreation	TCM	Commuters: 2.5 (winter)–3 (summer)
Marinelli and Romano (1987)	Foresta Umbra (FG), forest recreation	TCM	0.517
Tosi (1987)	Cansiglio (BL)	TCM CVM	3.65 WTP: 12.5
Tosi (1987)	Val Calamento (TN)	TCM CVM	4.15 WTP: 1.1
Tosi (1987)	Val Cimoliana (PN)	TCM CVM	5.7 WTP: 1.1
Frigo (1988)	Dolomiti bellunesi (BL)	TCM CVM	0.95 1.525
Gatto (1988)	Dolomiti bellunesi (BL)	TCM CVM	0.81 1.28
Gatto (1988)	Abetina Reale (RE), outdoor recreation	TCM CVM	1.165 1.32
Romano (1989)	Orecchiella Natural Park (LU), outdoor recreation	TCM CVM	3.275–6.038 WTP: 8.976
Signorello (1990)	Preservation of the 'Oasi del Simeto' (CT), natural reserve	CVM-DC	10.69 (median) 12.135 (mean)
Venzi and Rivetti (1990)	Giardino di Ninfa (LT), outdoor recreation	TCM	16.25
Cesaro and Merlo (1991)	Bacino Misa (AN)	TCM CVM	5.925 WTP: 4.9

Table 11.7 (continued)

Authors	Site	Method	Results (in current Euro)
Cesaro and Merlo (1991)	Rio Novella (TN)	TCM CVM	9.225 WTP: 1.375 WTA: 11.75
Romano and Rossi (1994)	Tuscan Appennines (AR), trekking	TCM CVM-IBG	4.578 16.685
Signorello (1994)	Valle dell'Anapo Natural Reserve (SR), outdoor recreation	CVM-DC	3.72 (mean); 3.405 (median)

Source: Adapted from Romano and Viganò (1995) and Merlo (1996).

In a CVM study carried out in the *Netherlands* in 1987 (Linden and Oosterhuis, 1988)[3] with the aid of postal questionnaires people were asked to state their WTP for the conservation of Dutch forest and heathland under the assumption that 80 per cent of forests and 90 per cent of heathland would be severely damaged by acid deposition by the year 2010. The survey showed that WTP was almost Dfl 23 (about 10 Euro) per household per month. Using an aggregation procedure the WTP of the Dutch population was calculated to be Euro 650 million per year. Although the underlying scenario does not seem realistic any more given present levels of pollution and the present level of knowledge on the effects of acid deposition, the study still shows the concern of the Dutch population for their natural environment.

Valuation of forestry resources in the *United Kingdom* has been conducted in a number of locations. A number of different attributes have been valued, including recreational use. Techniques used include contingent valuation, travel cost and choice experiment (CE) approaches.

Environmental degradation has a detrimental impact on recreational use. However, the studies to date in the UK have focused not on environmental degradation, but on the total value derived from having access to the forest. Thus, it is difficult to measure the impacts on forest resources and their associated benefits of a change in environmental quality. Given the site-specific nature of the studies, and a lack of studies researching the impact of environmental degradation on forestry in the UK, no attempt has been made to quantify the impacts of environmental damage to forests in the UK.

A summary of some of the main studies which have attempted to value forests in the United Kingdom in recent years is presented in Table 11.8. All figures are in 1995 Euro.[4] As can be seen from the table, the values given to forestry resources are quite site-specific, and are dependent on issues such as access as well as the quality of the forest.

Studies have ranged in their scope from valuing national parks such as the Yorkshire Dales (Bateman et al., 1994) to valuing forestry access in a number of areas in the UK. In addition, Hanley et al. (1998) have conducted a choice experiment alongside contingent valuation to evaluate the value of an 'ideal forest'. The impacts of different characteristics of forests were valued by Hanley and Ruffell (1991), who found that tree height diversity, the coverage of broadleaf trees and water features in forests were all significant in increasing the value attributed to an area of forest.

11.2.5 Defensive Expenditures for Forest Areas

Defensive expenditures for forest areas could only be identified for Germany. Kroth et al. (1989) quantified the costs of all mitigating measures in the affected West German forest areas to be between Euro 41.2 million and 112.9 million per year in the period from 1988 until 1992. Defensive expenditure for nature protection must – ideally – be calculated as net costs, net of natural services, that is, benefits obtained from healthy forests. For example, one hectare of spruce forest can take up to 42 t of dust per year. Using filters would cost DM 42 000, which corresponds to about 21 000 Euro (*Frankfurter Allgemeine Zeitung*, 1995).

11.3 OTHER ECOSYSTEMS AND BIODIVERSITY

Previous studies, as presented in Markandya and Pavan (1999, pp. 92–101), gave a general overview of valuation impacts on ecosystems, but did not aim to achieve a common valuation methodology for the countries under consideration. This publication will provide an update to the existing valuation literature in this area and try to apply a common approach to restoration costs wherever possible. Considerable progress in the area of restoration costs could be made through two projects that have been undertaken: one project was commissioned by the German Federal Environmental Agency on the role of biodiversity for a sustainable economy (see Geisendorf et al., 1996), the other is the long-term nature restoration programme of the Dutch government, called the Ecological Main Network, with a number of studies for assessing its costs.

Table 11.8 Summary of recent UK forestry valuation studies

Location	Study	Method	Value (1995 Euro)	Payment vehicle
Wantage, Oxfordshire	Bateman et al. (1996)	CVM	13.38 per household per annum 1.10 per adult visit	Annual payment Per visit, car parking
Derwent Country Park, Gateshead	Bishop (1992)	CVM	1.38 max. WTP per visit 26.42 per year	Per visit entrance charge Annual payment
Whippendell Wood, Watford	Bishop (1992)	CVM	1.91 max. WTP per visit 38.54 per annum	Per visit entrance charge Annual payment
Eight sites in UK	Willis and Benson (1989)	NS	3.32 per visitor	Consumer surplus for recreation
New Forest	Willis and Benson (1989)	NS	2.41 per visitor	Consumer surplus for recreation
Loch Awe	Willis and Benson (1989)	NS	5.57 per visitor	Consumer surplus for recreation
Yorkshire Dales (1)	Bateman et al. (1994)(2)	CVM	32.49 per resident household per annum (1996) 27.60 per visiting household per annum	Annual payment to preserve landscape
'Ideal forest'	Hanley et al. (1998)(2)	CE	42.13 per household per annum	Maximum value based on choice experiment. Used photographs to ask people to rank different choices
		CVM	32.20 per household per annum	

Table 11.8 (continued)

Location	Study	Method	Value (1995 Euro)	Payment vehicle
Lynford Stag	Brainard et al. (1999)	TCM	2.38 per person per trip	Based on benefit transfer
60 sites in UK	Hanley and Ruffell (1991)	CVM	Av. 1.25 per trip, impact of tree height diversity: 0.44, broadleaf trees: 0.66 and water area: 0.93	Payment card

Notes:
(1) Value is for whole national park, not just woodland or forestry.
(2) No date stated in paper for prices, so prices adjusted for year of publication of paper

Abbreviations
CVM Contingent Valuation Method
CE Choice Experiment
HPM Hedonic Price Method
TCM Travel Cost Method
NS Not stated

11.3.1 Physical Damage Indicators

With respect to nature it is difficult, if not impossible, to determine a reference against which to compare the current situation, parallel to the use of a background concentration of pollutants in assessing current damage. To define the reference situation for nature as a situation without anthropogenic interference is somewhat curious as man has always interfered with nature in large parts of Europe. In *Germany* there is a lack of national comparative data on biodiversity based on a monitoring system. Existing data are available only in terms of endangered species and biotopes. In 1997 the Federal Statistical Office initiated a new statistical approach for the registration of structural characteristics of landscapes and ecosystems. It is close to the British 'countryside survey' (see Barr et al., 1993) and is called an ecological spot check of sites. This new structural information allows for an immediate description of biodiversity (for example, number of species in a certain year) and also quantifies important elements of influence on biodiversity such as the percentage of sites with a variety of use intensities (Hoffmann-Kroll et al., 1998). Up to now there has been a pilot study in the agricultural areas of the states of

Brandenburg, Berlin and Thuringia. It is planned to extend the ecological spot checks to the national level and to apply them to areas of permanent observation which are not only nature protection sites, but also other sites. The approach of the Federal Statistical Office is restricted in its definition of biodiversity, in its sense of diversity of ecosystems and landscapes as well as diversity of species. Genetic diversity is excluded. Table 11.9 illustrates the classification used for the description of ecosystems and landscapes. The method of the ecological spot check applies 27 types of ecosystems and more than 500 different biotopes. Separate analysis of flora and fauna in their respective habitats is undertaken. The spot check was realised using ARC/INFO. As a result of the ecological spot checks indicators of biodiversity will be built. In terms of landscape three different topics will be described: intensity of use, structural diversity and rarity.

As far as biotopes are concerned, on average five indicators are used. For arable land these five indicators are: soil composition, the existence and size of field hems, slope of arable land and vegetation with wild herbs. Some relevant indicators for the agricultural area are shown in Table 11.10 as an example of an application.

The results of the pilot study show that the suggested indicators are appropriate and the necessary data can be collected without entailing excessive costs. For the tested areas the quality of biotopes could be assessed with relative accuracy.

In June 1998 a comprehensive red list of endangered animal species was published for the first time for the whole of Germany. In comparison to the earlier version from 1984 the observation area has been vastly extended through reunification. Of the total of 45 000 animal species 16 000 have been evaluated in terms of their endangered status. About one-third of the 100 different species of mammals in Germany are endangered, less than a third of all butterflies, fish and nesting birds; with respect to ants and dragonflies every other species is endangered (see Table 11.11). Table 11.12 presents extinct and endangered species for flora.

Italy ratified the Convention on Biodiversity in 1994, two years after its approval in Rio de Janeiro. Following the ratification, Italy defined the 'Strategic Guidelines for the Implementation of the Convention on Biodiversity'. For the implementation of the Convention a survey was carried out in order to improve knowledge on biodiversity and results were collated using existing surveys. These surveys show that Italy has 5900 different species of flora, and more than 56 344 of fauna, of which 56 168 are invertebrates and 176 vertebrates. Approximately 1.3 per cent of these species are protected.

Table 11.9 Classification of types of ecosystems and biotopes

1	Wadden Sea
	10 Wadden areas and sandy areas
2	Waters
	20 Creeks and rivers
	21 Estuaries
	22 Flowing technical waters
	23 Lakes, ponds
	24 Non-flowing technical waters
	25 Springs
3	Agricultural types of ecosystems
	30 Arable
	31 Vineyards
	32 Intensively wooded areas
	33 Meadows with fruit trees
	34 Inland grassland biotopes
4	Forest types of ecosystems
	41 Deciduous forests
	42 Evergreen forests
5	Other natural types of ecosystems
	50 Beaches, dunes, coasts
	51 Coastal and mountain grasslands
	52 Bushy areas
	53 Areas with small shrubs
	54 Moors
	55 Reeds
	56 Caves
	57 Cliffs, areas of poor vegetation
	58 Single trees, avenues
	59 Hedges
6	Other technical types of ecosystems
	60 Quarrying and mining areas
	61 Settlements
	62 Technical types of biotopes

Source: Hoffmann-Kroll et al. (1998, p. 63).

Table 11.10 Selected indicators for landscape quality in agricultural areas in Germany

Overall topic	Specific indicator	Basic data to be collected	Additional information	Indicator
Intensity of use	e.g. Degree of natural state	Areas of different biotopes in ha	Degree of natural state in four grades	% of areas of nature and close to natural biotopes
Structural diversity	e.g. Species diversity for selected species	All bird species		Average no. of species per km^2
		All butterfly species		Average no. of species per km^2
Rarity	Existence of rare and endangered biotopes	Areas of different biotopes in ha	Degree of endangeredness of all biotopes in five grades according to the 'red list'	% of areas of endangered types of biotopes according to the 'red list'

Source: Hoffmann-Kroll et al. (1998, p. 67), authors' selection of indicators.

Whereas a 'red list' of endangered species has been compiled for several years for mammals and birds, a lack of knowledge exists for other species, especially invertebrates. According to this list, in Italy 125 species of birds are endangered, as well as 136 species among mammals, reptiles, amphibians and fish.

In 1992 the Insitute of Entomology at the University of Pavia estimated for the first time the number of endangered invertebrates according to different criteria, as shown in Table 11.13.

At European level one of the most important and comprehensive laws on biodiversity is the so-called 'Habitat Directive' (Directive 92/43/CEE). This constitutes the most recent tool to protect habitats and endangered species. This Directive together with the Convention of Rio consolidated the idea that natural habitats need to be preserved in order to protect endangered species, following the idea introduced at European level by Directive 79/409/CEE for the protection of birds. According to these Directives, it was decided to set up special conservation areas and special protection areas for birds, which constitute the so-called 'Nature 2000 network'.

Table 11.11 Red list of endangered animals in Germany

Species	No. of species examined	Endangered species*	
		Absolute	% share
Mammals	100	33	33
Nesting birds	256	70	27
Reptiles	14	11	79
Amphibians	21	13	62
Fish	257	66	26
Flies	428	149	35
Butterflies	1450	451	31
Bees	547	237	43
Ants	108	59	55
Beetles	108	59	55
Dragonflies	80	44	55

Note: * The term 'endangered species' includes the red list categories 'endangered', 'strongly endangered' and 'threatened by extinction'.

Source: Bundesamt für Naturschutz (1998).

Table 11.12 Endangered flora in the FRG

Organism	No. of extinct species (% of all species)	No. (%) of currently endangered species
Fern and flowering plants	60 (2%)	637 (26%)
Moss	15 (2%)	84 (8%)
Lichen	26 (1%)	380 (21%)

Source: Plachter (1991).

In Italy, protected areas at present cover 6.58 per cent of the total national surface, that is, 1 981 287 hectares, which is still below the threshold of 10 per cent called for at international level. Protected areas include national parks, as well as national and regional protected areas.

The situation in the *Netherlands* is extreme, as the Netherlands are to a large extent created by man. Notwithstanding this methodological difficulty, it is always meaningful to compare the current situation of nature with the situation in less industrialised times. Table 11.14 provides some illustrative figures on the decline of specific biotopes in the Netherlands

Table 11.13 Endangered invertebrate species

Species	No.	%
Rare	638	26.20
Endemic	620	25.46
Rare, endemic	252	10.35
Very rare	211	8.67
Very rare, endemic, vulnerable	107	4.39
Endangered	95	3.90
Rare, vulnerable	92	3.78
Endemic, endangered	72	2.96
Very rare, endemic	63	2.59
Vulnerable	61	2.51
Rare, endangered	57	2.34
Partly endangered	52	2.14
Endemic, vulnerable	33	1.36
Very rare, endemic, vulnerable	22	0.90
Rare, endemic, vulnerable	18	0.74
Very rare, vulnerable	17	0.70
Very rare, endangered	15	0.62
Rare, endemic, endangered	6	0.25
Extinct	4	0.16
Total	2435	100.00

Source: Ministry of Environment (1997).

between 1900 and today, and indicates the prevailing causes according to expert judgements.

It has been estimated that there are about 36 000 species of multicellular organisms in the Netherlands. Since 1900, about 13 per cent of water-based species and 4 per cent of land-based species have become extinct. There are still many species that are threatened with extinction, although in recent years the rate of extinction seems to have declined (RIVM et al., 1997).

Many plant species have declined, especially rare species and species that are characteristic of wet, nutrient-poor habitats. Certain bird species are declining, especially birds characteristic of marshes. The position of butterfly species is a reason for concern. Table 11.15 lists the extent to which certain species in the Netherlands are threatened or extinct.

In the *United Kingdom*, as for other nations, problems with quantifying changes in semi-natural ecosystems make the identification of cause and effect relationships tenuous. At a broad level, the land cover of Great Britain has been monitored since the 1930s, initially with the Land

Table 11.14 Decline of biotopes in the Netherlands since 1900, and dominant causes

Biotope	Decline since 1990 (%)	Dominant cause
Living bog	>95	Reclamation, nitrogen deposition
Dry nutrient-poor grasslands	c. 95	Nitrogen deposition
Wet nutrient-poor grasslands	95	Nitrogen deposition, dehydration, reclamation
Heathland	90	Reclamation, nitrogen deposition, afforestation
Sand drifts	80	Cultivation, succession, nitrogen deposition
Stagnant bog	70	Reclamation, dehydration
Tidal salt marshes	50	Impoldering, cultivation
Marshes	40	Reclamation, dehydration, succession
Hedgerows	35	Land division
Dunes	15	Urban development
Forests	*increase* of 32%	Afforestation

Sources: RIVM (1996, Table 5.1.1, p. 278), and RIVM et al. (1997, p. 24).

Utilisation Surveys (Stamp, 1947; Coleman, 1961), then with the Monitoring Landscape Change project (Hunting Technical Surveys, 1986) and more recently with the Countryside Surveys (Barr et al., 1986, 1993). It is only the Countryside Surveys, which started in 1978, that have attempted to record ecological change in detail.

Changes in broad land cover categories usually reflect positive management decisions such as agricultural improvement, urbanisation and afforestation while the consequences of atmospheric pollutants are more likely to be seen in changes in quality of habitats and ecosystems. There is a strong interaction between the two measures both ecologically and economically. Reduction in the area of a cover type is expected to reduce the number of different species found there (MacArthur and Wilson, 1967) and increasing rarity generally increases the economic value placed on organisms and habitats. Those values can be measured in a number of ways, directly through money spent on conservation and protection to less direct methods such as willingness to pay.

Table 11.16 shows the changes in semi-natural cover types, set in the context of the other changes in land cover in Great Britain between 1984 and 1990. The major increase in semi-natural cover types is in rough grass. This category includes set-aside and agricultural grasslands, which

Table 11.15 Number and sensitivity of certain species in the Netherlands

Species	Total no. of species	Extinct	Severely threatened	Threatened	Potentially threatened	Not threatened
Birds (nesting)	172	3	10	23	21	115
Mammals	60	4	1	3	17	35
Reptiles and amphibians	23	0	2	4	9	8
Fungi	2475	202	253	289	911	820
Butterflies	70	18	7	11	12	22

Source: RIVM et al. (1997).

are being left derelict and converting, in the short term, into areas of tall herb.

At a more detailed level, the Countryside Surveys have shown changes in the occurrence and frequency of groups of species between 1978 and 1990. Rather than presenting individual species, the data are aggregated into groups of species with similar environmental requirements which are consequently often found together (Bunce et al., forthcoming a). Table 11.17 shows the changes in the average number of species found in areas surveyed in 1978 and revisited in 1990. Most of the changes show a decrease in the number of species and in half of the classes the changes are statistically significant. Like the category rough grass in Table 11.16, the tall grassland/ herb class shows an increase (although not significant). There is a sharp contrast between the changes in vegetation in upland and lowland woods. The lowland woods show a small increase in the number of species whereas the upland show a marked decline. The reason probably reflects the history and geographic differences in planting, the uplands having larger areas of commercial planting with predominantly non-native conifers. Once the canopy of a coniferous wood closes, the ground flora declines and may totally disappear.

Even using this data, it is not possible to prove a cause and effect relationship between species loss and atmospheric pollution at a broad national level. If the characteristics of the species in the quadrats are examined it may be possible to show a trend towards species considered to be tolerant of increased levels of different pollutants. English Nature, the UK agency responsible for promoting nature conservation, has identified those species considered to be indicators of unimproved grassland. By selecting quadrats containing species which are indicators of unimproved calcareous, acid and mesotrophic grassland some measure of the direction of change may be

Table 11.16 Land cover changes in Great Britain between 1984 and 1990

Cover type	Stock 1984		Change 1984–90	
	Area	*% GB*	*Area*	*% change*
Agriculture	1218	53	−38	−3
Forestry	254	11	8	3
Urban	198	9	8	4
Water	36	<1	–	−1
Semi-natural				
Rough grass	66	3	29	45
Moorland grass	128	6	−4	−3
Open heath	128	6	–	–
Dense heath	47	2	+	1
Bogs	159	7	1	1
Saltmarsh	4	<1	–	−9
Bracken	42	2	−5	−11

Note: Areas are presented in $km^2 \times 10^2$; positive values represent gains, negative losses; – and + indicate changes of less than 1 in terms of losses and gains respectively.

Source: Barr et al. (1993).

identified. However, the results (Table 11.18) show significant declines in both mesotrophic and acidophilic quadrats, once again demonstrating the complexity of factors determining change and the weakness of a correlative approach.

A recent project studying the changes recorded in the Countryside Surveys and identifying factors which could contribute to changes in biodiversity showed that there is a widespread trend towards the simplification of natural habitats with a few generalist species becoming abundant and reducing diversity (Bunce et al., forthcoming b). The study showed the way changes in vegetation affect bird, butterfly and bee populations. It illustrates how butterfly food plants are generally declining across all of the aggregate vegetation types. Birds and butterflies are emotive creatures and people are generally willing to pay for their conservation. Unfortunately not all butterflies show the same behaviour pattern, and obvious, showy species such as the peacock (*Inachis io*) are associated with the more widespread generalist plants, giving a false impression that butterfly numbers are up. The evidence suggests that butterfly diversity is under serious threat.

In order to try and show a causal link between atmospheric pollutants and loss of species, a more in-depth study of individual species was carried

Table 11.17 Changes in average numbers of plant species found in 'quadrats' in 1978 and 1990

Aggregate class	No. of species 1978	Change in number 1978–1990	Change %	Significance
Crops/weeds	6.8	−1.5	−22	**
Tall grassland/herb	13.3	0.5	4	ns
Fertile grasslands	12.5	−0.5	−4	ns
Infertile grasslands	21.2	−2.9	−14	**
Lowland woods	12.5	0.2	2	ns
Upland woods	20.4	−4.3	−21	**
Moorland/grass mosaic	22.1	−1.4	−6	ns
Heath/bog	17.6	1.0	6	*

Note: Each quadrat was classified into one of eight aggregate classes and only those quadrats which did not change class were counted. The significance is estimated using a t-test, where * $p < 0.05$; ** $p < 0.01$; ns = not significant.

Table 11.18 Changes in number of quadrats containing at least one species used as an indicator of different types of grassland by English Nature

Indicator species	No. of plots in 1978	No. of plots in 1990	Significance
Calcicolic species	255	276	ns
Mesotrophic species	1156	1068	**
Acidophilic species	1243	1189	**

Note: The significance is estimated using a chi-square test, where * $p < 0.05$; ** $p < 0.01$; ns = not significant.

out by Ecofact. Sampling across northern Britain of *Racomitrium lanuginosum* (the species is only found at moderate altitude), a moss that has been reported to be intolerant of high levels of nitrogen deposition (Baddeley et al., 1994) showed occurrence where deposition levels were lower. However, examination of the nitrogen content of plants found did not show a strong relationship to modelled deposition. Even when the physiology of a single plant species is taken into account (Baddeley, 1991), it is not possible to categorically estimate the change in occurrence due to pollution.

11.3.2 Selected Studies on Economic Valuation of Ecosystems and Biodiversity

Whereas valuation studies on forests (see Section 11.2.4) capture the use value of these resources (basically recreational), this second part of the present chapter deals with the valuation of other ecosystems not directly associated with recreational functions and biological diversity. These types of valuation reflect non-use values to a large extent. As Solow et al. (1993) pointed out, the efficient allocation of biological resources conservation needs also to look at the preservation of diversity. This measure, however, does not only depend on its genetic distinctiveness, but also on the effect that its extinction would have on the extinction probabilities of other species.

In the following some CVM studies carried out at the European level will be presented.[5] Comparison of their results shows that they tend to converge, but without being identical. Using comparable questions, almost all the studies show a comparable order of magnitude for the willingness-to-pay for the recreational value of forests (see Table 11.19 and Geisendorf et al., 1996).

In *Germany*, the willingness-to-pay seems to reach the level of membership fees for environmental organisations. Most of the studies show mean values of about Euro 40 to 100 per household per year. A cautious calculation (for instance taking a value of Euro 40 and only 35 million households) results in a willingness-to-pay of about 1.4 billion Euro.

Apart from questions about the accuracy of these data and its consistency with axioms of microeconomic theory, it is important to agree on the fundamental and qualitative information which is inherent in the data. A specific, political interpretation which is more or less neutral to methodological difficulties concerning preference analysis, encompasses the following points:

- The vast majority of those interviewed for the studies recognise the conservation of biodiversity as an important task.
- In the opinion of the interviewees the use of financial means is appropriate for the conservation of biodiversity.
- Interviewees are willing to participate in the exercise.
- Their willingness-to-pay can be interpreted as ‘fair’. Neither income limitations are overemphasised nor exaggerated altruism is demonstrated (Geisendorf et al. 1998).

Other *Dutch studies* (Table 11.20) have consistently found a positive willingness to pay for specific landscapes and ecosystems, for example the

Table 11.19 Survey on international and German valuation studies concerning biotopes

Authors	Method	Results
Valuation studies on single biotopes		
Willis, 1990 Conservation of three biotopes in northern England	TCM	£1.02 per visit £1.15 per visit £2.29 per visit
Hanley and Craig, 1991 Conservation of Scottish highlands	CVM	£16.79 per person, one time payment (Flow Country)
Cobbing and Slee, 1993	CVM	£26.64 per person, one time payment (Mar Lodge Estate)
Valuation studies on the conservation of rural landscapes		
Corell, 1994 Rural landscape in the German Lahn-Dill-Bergland	CVM	DM 17.13 per month
Zimmer, 1994 Nature protection and landscape management in German Emsland and Werra-Meißner-Kreis	CVM	DM 14 per household per month (fee for landscape management) and DM 2.50 per person per day (fee for spa location)
Valuation studies on nature protection in general		
Hampicke, 1991 Species and nature protection in Germany	CVM	c. DM 20 per household and month DM 3–7.5 billion per year (total)
Alvensleben and Schleyerbach, 1994 15% of Schleswig-Holstein as nature protection area	CVM	DM 16–25 per person per month

Source: Geisendorf et al. (1996, p. 26).

Wadden Sea (Wierstra, 1996; Spaninks et al., 1996) and natural agricultural landscapes (Spaninks, 1993; Brouwer, 1995). Two studies on the impact of natural amenities on house prices found significant positive correlations for many sites, though not all (Fennema, 1995; Luttik and Zijlstra, 1997). A recent study found a surprisingly close correlation between the level of willingness to pay of individuals for specific coastal ecosystems and the 'ecological' values assigned to these ecosystems by natural science experts (Ruijgrok, 1997). A recent, large telephone survey found that a clear majority (70 per cent) of people who occasionally visit forests would be willing to pay *at least ƒ* 1 (Euro 0.44) for such a visit (Foekema, 1997).

Analysis of the various studies makes two things very clear: (1) there is a general and consistent willingness to pay for the conservation of nature in the Netherlands; and (2) the amounts of money that individuals are willing to pay are highly context-specific, depending both on the specifics of the object of valuation and on the valuation context itself (see also Hoevenagel, 1994; Wierstra, 1996). It is therefore not possible to determine *the* value of nature in the Netherlands on the basis of these results, let alone the monetary value of changes in the quantity or quality of nature due to economic activities in a specific area.

11.3.3 Restoration Costs of Biodiversity

In their assessment of the socio-economic value of nature and biodiversity in *Germany*, Geisendorf et al. (1996) calculate the costs of restoring nature to a degree that would provide a minimum level of protection to Germany's biodiversity. The restoration programme basically consists of the extensification of 10 to 15 per cent of Germany's agricultural area, and some other measures to realise a sufficiently large, coherent and integrated semi-natural area. Geisendorf et al. (1998) estimate the annual costs of this programme to be of the order of magnitude of DM 2 billion (Euro 1 billion). According to these calculations the extensive use of agricultural land (for example, through extensive breeding) would be economically feasible in the long run when there are annual payments to the producers of between DM 300 and 1000 per hectare per year. There is a clear frequency of values of between DM 600 and 700 per hectare per year. These payments must be understood as additional to the revenues resulting from the sale of products and are only valid for locations where there is no incentive for intensive agriculture. Usually these locations are well-suited to extensive use.

In the assessment of nature and biodiversity in *the Netherlands* in the sections above, it was pointed out that nature in the Netherlands is highly

Table 11.20 Selected Dutch nature valuation studies (1987–97)

Topic	Area	Year	Method[a]	WTP[b]	Remarks	Author
Conservation of forests and heathland	Netherlands	1987	CVM	*f*22.83 per household per month	Mail survey; n = 459[c]	Linden, Van der. and Oosterhuis (1988)
Nature and landscape damage due to eutrophication	Netherlands	1989	BT	*f*60–315 Mio. per year		Baan and Hopstaken (1989)
Restoration of peat meadow region	Peat meadow region	1989	CVM	*f*16–46 per person per year	Experiment with students, n = 85	Hoevenagel (1994)
Restoring woodlands to heathland	Netherlands	1989	CVM	*f*27.30 per person per month	Experiment with students, n = 288	Hoevenagel (1994)
Conservation value of the Wadden Sea	Wadden Sea	1992	BT	US$600 per hectare	Based on the conservation costs of a marsh area in Florida (US) in the late 1960s	De Groot, (1992)
Nature conservation in agricultural areas	Peat meadow area in Friesland (500 ha)	1993	CVM	*f*8.61 per household per month	Mail survey; n = 146	Spaninks (1993)
Value of a new forest (1000 ha)	South-Holland	1993	CVM	*f*71.74 per household (one-time payment)	Mail survey; n = 684	Verkoijen (1994)
Conservation of small uninhabited island in Wadden Sea	Rottumeroog	1994	CVM	*f*12.18 per household per month	Face-to-face; n = 225	Wierstra (1996)
Nature conservation in agricultural areas	Peat meadow area in South-Holland (15 660 ha)	1994	CVM	*f*28.70 to *f*72.10 per household per year	Mail survey; n = 412	Brouwer (1995)
Impact of natural amenities on house prices	Apeldoorn	1995	HP	Increase on house price of 6% and 8%[d]		Fennema (1995)

Table 11.20 (continued)

Topic	Area	Year	Method[a]	WTP[b]	Remarks	Author
Financial valuation of nature	Netherlands	1996	MV	*ƒ*87 mln	Revenues from hunting licences and fishing rights	Barris and Pommée (1996)
Impact of natural amenities on house prices	Various locations	1997	HP	Various[e]		Luttik and Zijlstra (1997)
Sustainable development of the Wadden Sea	Wadden Sea	1996	CVM	*ƒ*70.71 per household per year	Mail survey; n = 544	Spaninks et al. (1996)
Use value of forests	Netherlands	1997		[f]	Telephone survey; n = 1031	Foekema (1997)
Value of coastal ecosystems	Coastal locations	1997	CVM	*ƒ*1.93 to *ƒ*4.50 per visit [g]	Face-to-face; n = 332	Ruijgrok, (1997)

Notes:

a Contingent Valuation Method (CVM); Benefit Transfer (BT); Market Value (MV); HP (Hedonic Price Method).

b *ƒ* stands for Dutch Guilders.

c n = usable sample size.

d 6 per cent if house is in walking distance from green scenery; 8 per cent if house has a view over green scenery.

e The study examined the effect of different types of green scenery on the prices of new houses (built after 1970) in 8 municipalities. The results are location-specific: a view over green scenery raised house prices by 4 to 8 per cent in 3 locations, but had no effect in 6 others; nearby water raised house prices everywhere (5 to 12 per cent); green scenery bordering the neighbourhood raised house prices in one location (12 per cent), but had no effect in three others; water bordering the neighbourhood had effect in all locations (5 to 12 per cent); an attractive, green environment in the wider setting had a positive effect in all locations (5 to 10 per cent).

f 70 per cent of those respondents who occasionally visit forests are at least willing to pay an entrance fee of *ƒ* 1 for a visit to a forest.

g From *ƒ*1.93 for a visit to a half natural tidal area to *ƒ*4.50 for a visit to an almost natural dune landscape.

fragmented and that reclamation and development of natural biotopes have been among the dominant causes of nature and biodiversity losses. A nature restoration programme in the Netherlands would thus include the expansion of nature's territory and the integration of the at present fragmented protected natural areas. Such a programme indeed exists in the

Netherlands: it is government's long-term nature restoration programme, known as the Ecological Main Network. A number of studies have been carried out to assess its costs.

The Ecological Main Network (to be called EHS hereafter, the abbreviation of its Dutch name 'Ecologische Hoofdstructuur') is the long-term nature protection and development programme of the Dutch government. Its objective is to create an integrated network of sustainably protected ecosystems of national and international importance. The enlargement and linking of natural areas is meant to diminish nature's vulnerability to external influences. The EHS should enlarge nature's territory from about 500 000 ha at present to about 700 000 ha in the year 2018. It will do so by nature development, the creation of additional nature reserves and by management agreements with farmers (RIVM et al., 1997, p. 62).

The costs of creating the EHS have been assessed by a number of authors (Oskam, 1994; Slangen, 1994; Sijtsma and Strijker, 1995). Sijtsma and Strijker (1995) assess the costs of nature development and the creation of nature reserves and management-agreements. Oskam (1994) and Slangen (1994) only assess the costs of nature development and the creation of nature reserves. The three studies all make a distinction between financial costs for the government and social opportunity costs.

Financial and social opportunity costs for the Dutch case

The main financial costs concern the acquisition of agricultural land to be converted to nature reserves, the management of nature reserves and the financial transfers to farmers involved in management agreements. For the EHS 144 000 ha of agricultural land will be transformed into nature reserves and 100000 ha will be placed under management agreement. Sijtsma and Strijker (1995) use an average price of agricultural land of about Euro 18 000 per hectare. The management costs of nature reserves are Euro 360 per year, of which Euro 186 are financed by the government. The average subsidies for management agreements are Euro 224 per hectare for 'light' management regimes and Euro 493 per hectare for 'heavy' management regimes. The average subsidy is Euro 358 per hectare. The total discounted financial costs for the government of the acquisition of land, the management of nature reserves and the management agreements in the EHS is then Euro 2.3 billion (Sijtsma and Strijker, 1995).[6] Slangen and Oskam calculate higher financial burdens. Without management agreements (Euro 0.45 billion in Sijtsma and Strijker), Slangen estimates the total discounted financial costs at 2.5 billion, and Oskam presents a figure of Euro 3.8 billion. If we add the financial costs of Euro 0.45 billion for management agreements to the estimates of Slangen and Oskam, their totals become Euro 3 billion and Euro 4.3 billion, respectively. In the three studies, the annual financial costs for the government of realising the EHS are between Euro 106 million and Euro 198 million.

The creation of restoration programmes like the EHS takes up scarce resources, thereby generating social opportunity costs. These costs are difficult to assess. What is the opportunity cost of the lost agricultural production in highly distorted agricultural markets? Can idle resources (that is, agricultural labour) be employed elsewhere in the economy? Do the financial transfers involved in the acquisition of land have negative repercussions on the economy (excess burden)? These are difficult macroeconomic questions. Despite these difficulties, the three studies have produced estimates of the social opportunity costs. Sijtsma and Strijker estimate these costs at Euro 1.5 billion, Slangen estimates costs of Euro 3.5 billion and Oskam estimates costs of Euro 3.9 billion.[7] Annual social opportunity costs in the three studies are then Euro 70 million, Euro 201 million and Euro 183 million, respectively. It seems therefore safe to suggest that the annual social opportunity costs of realising the EHS will be somewhere between Euro 70 million and Euro 200 million.

11.4 CONCLUSIONS

At this stage a number of remarks are required from a methodological point of view (for policy conclusions see Section 16.6). The environmental damage costs of economic activities are assessed by estimates of individuals' willingness to pay to avoid this damage. The assessment of environmental damage costs through estimating restoration costs to achieve collectively decided objectives is a completely different approach. The restoration cost approach is the approach taken by Hueting (1989) in the calculation of his Sustainable National Income (see Chapter 2). As pointed out in Chapter 2, apart from theoretical difficulties with this approach, a major practical problem is the determination of sustainability norms. In terms of our study, what is the 'background' or 'reference' level of nature against which to measure the damage to nature in a particular year? Taking the collectively decided objectives of, for example, the EHS in the Netherlands as the background or reference level of nature is obviously open to challenge.

Nevertheless, given the methodological and practical problems involved in estimating reliable willingness-to-pay figures for the avoidance of damage to nature and biodiversity, the restoration cost approach, although second-best, is *in this case* probably the only way to monetise damage.

A second methodological issue involves the negative effects of environmental pollutants on the quality of nature and biodiversity. Table 11.7 indicated that the dominant causes of the decline of nature in the Netherlands (but also elsewhere) in this century involved both the direct conversion of nature (land reclamation) and indirect effects of pollution and (ground) water management. While for instance the Dutch EHS takes account of the direct causes and some indirect causes (for example, groundwater management), it does not take account of the deposition of pollutants on nature areas. For example, Figure 11.1 showed that the average level of nitrogen deposition still exceeds critical levels by a wide margin in most ecosystems in the Netherlands. The restoration of nature and biodiversity therefore also calls for a (major) reduction in environmental pollutants. This suggests that the true restoration costs of nature will be higher than the costs of the restoration programmes.

NOTES

1. This figure is derived from a synthesis report on forest condition in the new states of Germany (Kurth, 1991: S. 108).
2. Klein (1997).
3. We acknowledge that this procedure is likely to result in increased uncertainty over the aggregate value. First, multiplying the monthly figure by 12 is believed to create overestimation (as people do not think how much this would be on an annual basis) and secondly, the scenario presented to respondents seems to have been misinterpreted. Nevertheless, we include the estimate as a guide to WTP for the Netherlands, for which few studies were found.
4. These values have been converted using consumer price indices and at an exchange rate of 1.20 Euro.
5. For *Italy* the literature on CVM studies concerning biodiversity appears to be very sparse.
6. An interest rate of 4 per cent is used. This is the officially prescribed rate. Sijtsma and Strijker use a planning horizon of 50 years; Slangen uses a planning horizon of 30 years.
7. The estimates of Slangen and Oskam are inclusive of Euro 0.43 billion for management agreements.

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12. Valuation of water

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12.1 INTRODUCTION

The purpose of this study is to establish linkages between sources of environmental damage and concentration levels or load of pollutants, thereby enabling the identification of contributions to damage on a sectoral and geographical basis. This approach is, in principle, suitable for water pollution but its application appears more problematic in the case of water than with atmospheric pollution.

In this chapter we first develop a possible framework for the estimation of damages caused by water pollution. This uses, as a willingness-to-pay (WTP) proxy, the share of expenditure on water treatment that can be attributed to each industrial sector and to households on the basis of the relative contribution to water pollution by each sector. Whilst making the link between source and damage, this approach has not yet yielded monetary values since data on water treatment expenditure are scarce. We then present an initial exercise in the valuation of one type of damage using WTP techniques as a case study for the type of work which could be developed in the future (that is, the case of angling in the UK).

Although synthesis of these two approaches is clearly possible we present them as separate elements in this chapter since we do not yet have sufficient country data to implement this synthesis. In both approaches, the application is best carried out at the river catchment level in order to consider all the aspects influencing water quality in a specified area.

The methodology used for this analysis suggests that the following stages should be undertaken in its application to freshwater:

1. estimation of potential pollutant loads, that is, loads produced by industrial activities, directly discharged to the environment or collected in sewerage, for each economic sector and region;
2. calculation of effective pollutant loads, that is, loads that reach water bodies after processes such as wastewater treatment, soil filtration and chemical transformation;

3. calculation of pollutants concentrations in water bodies, through mass balances in water bodies;
4. calculation of the impact of water pollution on different receptors (population, agriculture, natural biota and so on), using data about water uptakes, water treatment, receptor distribution, water consumption and exposure–response functions giving correlation between concentrations and impacts;
5. calculation of monetary damage, using monetary values for each of the impacts calculated during the previous phase;
6. evaluation of water treatment costs.

However there are significant complicating factors that increase the difficulty of applying this methodology, as explained below.

- In stage 1, the existing techniques for estimating water pollutants loads are much more uncertain than those for atmospheric pollution, and they refer to a very low number of parameters, compared with the wide range of parameters of concern when considering water quality. Studies are available in Italy for BOD (Biological Oxygen Demand) and COD (Chemical Oxygen Demand) loads from civil and industrial sources (depending on different industrial sectors), while for N and P the estimate is based on civil discharges, from which the industrial values are derived. Some studies also exist for nutrients coming from agriculture and livestock. A study in the UK examined the consequences of increased phosphorus, predominantly from sewage, into a lake and attempted to cost the preventative measures required.
- These techniques allow the calculation of potential loads, that is, pollutants directly emitted by industrial or civil sources. Only a fraction of these loads reach water bodies (effective loads) after being treated, filtered by the soil and chemically transformed. In order to account for these processes (stage 2), one should provide a precise description of river network, soil composition and permeability, and collect data on levels of water treatment. This is much more complex than for atmospheric dispersion, where the use of emission factors based on combustion processes and average atmospheric parameters to model atmospheric processes can produce sound results. Moreover, surface water and groundwater should be considered separately, which makes the problem even more complex.
- In order to calculate pollutant concentrations in water bodies, we need to calculate mass balance, starting from effective pollutant loads and river flows. Thus, more data are needed for the calculation, and again a detailed description of the river network is needed.

- The issue of water quality is substantially different from air quality. First, water can be used for different purposes, each one following different pathways, which should be considered separately. Moreover, the impact pathway for atmospheric pollution is somehow direct, because people cannot choose which air to breathe, or not to breathe (though people can, of course, move location). Similarly, crops and materials protection from atmospheric pollution is not feasible. Water is generally treated before consumption, thereby reducing some health damages. Thus, it is difficult to evaluate the population at risk of health damage from water contamination, and also to understand which part of water treatment costs are effectively caused by water pollution. For this purpose, studies at a very local level should be carried out, including epidemiological studies, contingent valuations, market analysis, and so on.

More simplified and – at the same time – more comprehensive methodologies need to be implemented in order to allow the valuation of physical or monetary damages from water pollution. In this chapter, physical damages are assessed through the calculation of a water quality indicator starting from measured parameter values (BOD_5). Monetary valuation consists in the attribution to different economic sectors, through the use of restoration costs (that is, water treatment costs) and the estimation of potential pollutant loads from different economic sectors.

12.2 THE MONETARY VALUATION OF WATER POLLUTION DAMAGES

The methodology described below assumes the costs of water treatment (for direct human consumption and for agricultural and industrial uses) as a proxy of the damage caused by water pollution. This is a first approximation only, for a number of reasons. First, this assumption neglects damages to the natural environment. Second, water for human consumption or some industrial purposes would, to some extent, be treated even if no pollution was present (treatment of hard water, turbidity removal and so on). Third, some residual damage can arise even after treatment, if some traces of pollutant are not eliminated.

It is to be noted that wastewater treatment costs are a proxy not for total damages but only for those damages that are restored in order to allow for human uses, be these civil, industrial or other. Taking water treatment costs as an approximation for valuation, these costs can be attributed to the different economic sectors (including households) according to estimates of the respective water pollutant discharge levels. For this purpose, an aggregated indicator of water pollution is needed, which can be calculated for all industrial and civil activities. One suitable parameter is BOD (Biochemical Oxygen Demand, that is, the oxygen needed by bacteria to convert organic matter into CO_2), on which data are relatively widely available, although it only measures organic pollution levels, neglecting other kinds of pollution (chemical, temperature and so on).

Potential loads estimation A methodology has been developed for the valuation of potential loads based on the number of people employed in each economic sector. For each sector, a relationship has been established between the number of workers employed and pollution in terms of BOD_5 (expressed as population equivalent). This is formulated relative to the ratio of aggregate BOD pollution against total resident population. The ratio, or coefficient, is approximately 6.94E-4 g/s BOD_5 per person. Expressed as people-equivalent, therefore, the coefficient for total resident population is 1. Table 12.1 below reports the coefficients for each industrial activity in

Table 12.1 Coefficients for the estimation of organic pollution of industrial activities, expressed as population equivalent per employee

NACE classification	Activity description	Coeff.
–	Resident population	1
C – Mining and Quarrying		
10	Mining of coal and lignite; extraction of peat	20
11	Extraction of crude petroleum and natural gas; service activities incidental to oil and gas extraction excluding surveying	30
12	Mining of uranium and thorium ores	0.6
13	Mining of metal ores	5
14	Other mining and quarrying	30
D – Manufacturing		
15	Manufacture of food products and beverages	98
15.1	Production, processing and preserving of meat and meat products	64
15.2	Processing and preserving of fish and fish products	31
15.3	Processing and preserving of fruit and vegetables	155
15.4	Manufacture of vegetable and animal oils and fats	230
15.5	Manufacture of dairy products	57
15.6	Manufacture of grain mill products, starches and starch products	1.5

Table 12.1 (continued)

NACE classification	Activity description	Coeff.
15.7	Manufacture of prepared animal feeds	24
15.8	Manufacture of other food products	24
15.9	Manufacture of beverages	483
16	Manufacture of tobacco products	7.5
17	Manufacture of textiles	17
18	Manufacture of wearing apparel; dressing and dyeing of fur	0.6
19	Tanning and dressing of leather; manufacture of luggage, handbags, saddlery, harness and footwear	17
20	Manufacture of wood and of products of wood and cork, except furniture; manufacture of articles of straw and plaiting materials	1.6
21	Manufacture of pulp, paper and paper products	118
22	Publishing, printing and reproduction of recorded media	0.6
23	Manufacture of coke, refined petroleum products and nuclear fuel	66
24	Manufacture of chemicals and chemical products	66
25	Manufacture of rubber and plastic products	10
26	Manufacture of other non-metallic mineral products	1.5
27	Manufacture of basic metals	2.3
28	Manufacture of fabricated metal products, except machinery and equipment	2
29	Manufacture of machinery and equipment n.e.c.	1
30	Manufacture of office machinery and computers	0.6
31	Manufacture of electrical machinery and apparatus n.e.c.	1
32	Manufacture of radio, television and communication equipment and apparatus	1
33	Manufacture of medical, precision and optical instruments, watches and clocks	0.6
34	Manufacture of motor vehicles, trailers and semitrailers	1.7
35	Manufacture of other transport equipment	1.7
36	Manufacture of furniture; manufacturing n.e.c.	1.7
37	Recycling	0.6
E – Electricity, Gas and Water Supply		
40	Electricity, gas, steam and hot water supply	1.4
41	Collection, purification and distribution of water	0.6

Note: n.e.c.: not elsewhere classified.

terms of population equivalent (a parameter that indicates the treatment capacity of a wastewater treatment plant in relation to the wastewaters produced on average by the inhabitant) per employee.

The advantage of this methodology is that data about the number of workers employed in each sector should be easily available. They are usually collected and published in periodical censuses of industry by national statistical offices. Moreover, the classification of the economic activities used is fully compatible with the NACE classification, adopted by Eurostat.

12.2.1 Data Availability

The data on the number of employees for each NACE sector, as well as the population, needed for pollutant loads calculation, could be collected during periodic censuses. Unfortunately, at present these data are only available free of charge in Germany and Italy. For the Netherlands, some data on water discharges have been collected, but a different classification, is applied, although based on people-equivalence. This is why Table 12.2 is presented only for Germany and Italy.

The pollution weighting for each industrial activity derived in Table 12.1 can be combined with the number of employees in each industrial activity to give the distribution of employees, and thus the contribution in terms of pollutant loads (expressed as percentages), by sector for Italy and Germany. This is summarised in Table 12.2.

Table 12.2 Number of employees and damage contributions by industrial sector

NACE classification	Population		BOD discharges (as a % of damage contribution)	
	Germany	Italy	Germany	Italy
Population	82 200 000	57 400 000	37.78	38.29
	Employees			
C Mining and Quarrying	180 007	48 561	1.85	0.94
10	137 481	1 012	1.26	0.01
11	6 280	9 736	0.09	0.19
12		0	0.00	0.00
13	320	1 496	0.00	0.00
14	35 926	36 317	0.50	0.73

Table 12.2 (continued)

NACE classification	Population		BOD discharges (as a % of damage contribution)	
	Germany	Italy	Germany	Italy
D Manufacturing	6 575 598	5 227 549	60.19	60.62
15	529 196	458 795	23.83	29.99
15.1	106 190	–	3.12	0.00
15.2	11 766	–	0.17	0.00
15.3	27 633	–	1.97	0.00
15.4	8 459	–	0.89	0.00
15.5	49 079	–	1.29	0.00
15.6	9 357	–	0.01	0.00
15.7	11 416	–	0.13	0.00
15.8	221 401	–	2.44	0.00
15.9	83 895	–	18.62	0.00
16	14 674	17 625	0.05	0.09
17	147 923	404 114	1.16	4.58
18	102 078	418 862	0.03	0.17
19	33 854	243 543	0.26	2.76
20	120 238	186 192	0.09	0.20
21	154 909	88 598	8.40	6.97
22	263 156	195 742	0.07	0.08
23	25 792	29 057	0.78	1.28
24	535 244	239 168	16.24	10.53
25	368 114	179 439	1.69	1.20
26	282 191	276 359	0.19	0.28
27	294 475	170 381	0.31	0.26
28	597 605	614 590	0.55	0.82
29	1 047 870	540 942	0.48	0.36
30	52 450	25 565	0.01	0.01
31	492 067	207 799	0.23	0.14
32	158 541	139 921	0.07	0.09
33	233 054	117 764	0.06	0.05
34	691 291	214 539	0.54	0.24
35	172 053	143 944	0.13	0.16
36	251 778	309 098	0.20	0.35
37	7 045	5 512	0.00	0.00
E Electricity, Gas and Water Supply	317 752	176 816	0.18	0.15
40	259 732	154 825	0.17	0.14
41	58 020	21 991	0.02	0.01
Total industry			62.22	61.71

The table shows that in both countries approximately 62 per cent of organic water pollution is generated by industrial sectors, the remaining 38 per cent being due to civil water discharges. Among the industrial sectors, the most polluting is food and beverages (30 and 24 per cent of the total in Germany and Italy respectively), followed by the chemical industry (16 and 11 per cent) and the paper industry (8 and 7 per cent). In Italy the textile sector is also quite important, accounting for 5 per cent of total organic pollution. In the Netherlands, pollution loads of 15 382, 12 625 and 5 630 people-equivalent are reported to be caused by consumers, producers and other polluters respectively. These percentages reflect the fact that only organic pollution is being considered. A more accurate approach should consider other pollution parameters, such as COD.

Data on water treatment expenditures are not readily available at the national level. Generally, available data only consider wastewater treatment expenditure, which, for the reasons highlighted above, are not appropriate for our purposes.

The situation of data collection on water treatment expenditure is different for the four countries:

- for Germany, only wastewater treatment costs are available at the moment, split into industrial sector and households;
- for the Netherlands, data on defensive expenditures borne by drinking water companies, and for horticulture, industry and commercial fishing have been collected;
- for Italy and the UK data only refer to wastewater treatment and moreover, they do not cover all sectors.

Accounting procedures for water treatment expenditure therefore need to be developed before the attribution of such expenditure to households and industrial sectors can be calculated on the basis of these reported percentages.

12.3 VALUATION OF WATER USING WILLINGNESS TO PAY ESTIMATES

This section reports on an initial attempt to apply the methodology of Impact Pathway Analysis to the valuation of water quality in the UK. We concern ourselves solely with the recreational and amenity value of water quality and produce first estimates of water quality value in terms of angling – specifically, coarse and game fishing. We also present an aggregate estimate of the amenity value of the presence of water reflected in house price variations. It was found that Impact Pathway Analysis of river water pollution was not sufficiently developed to be applied usefully. Therefore the methodology has effectively been truncated in order to use WTP values just for observed water damages.

The work builds on the valuation framework developed by the UK Environment Agency and adopted in the EC 'Guidelines for Cost-benefit Analysis of Proposed "Good Status of Water" Standards' report. It must be emphasised that the results given here are very preliminary. They are designed to stimulate discussion and give broad orders of magnitude for the damage costs resulting from river water quality deterioration from non-anthropogenically influenced levels.

12.3.1 Methodology

(a) Angling

The methodological framework that has been adopted for valuing recreational use applies WTP values to participation rates for given populations. In the case of angling, the analysis can be broken down by considering two scenarios that arise from a fall in water quality:

1. The impact of water quality is so severe that no angling takes place.
2. There is a reduction in the quality of the angling opportunity.

When the first scenario exists, the value of the total damages resulting from the polluting activity can be calculated using the following equation:

$$C_a = WTP_t \times T_n \times N_n \tag{12.1}$$

where C_a is the total annual cost under existing conditions, WTP_t is the willingness of each angler to pay for fishing as it is now, T_n is the number of angling trips per angler in a year and N_n is the number of anglers.

In the second scenario, the value of the total damages is calculated by the equation:

$$\Delta C_a = (WTP_t \times T_n \times N_n) - (WTP_{tm} \times T_m \times N_m) \tag{12.2}$$

The change in the annual total cost of angling due to a decrease in water quality (ΔC_a) can be calculated by subtracting the willingness to pay under new conditions from the cost under existing conditions (C_a). The willingness to pay under new conditions is calculated as the product of willingness to pay per trip (WTP_{tm}), the number of angling trips under modified conditions (T_m) and the numbers of anglers (N_m).

This methodology can, in theory, be applied to other recreational activities such as walking, bird-watching, canoeing and swimming. However, insufficient research has been carried out into the effects of water quality on the willingness to pay for these activities to enable useful estimates of these to be undertaken here.

There are a number of methodological and empirical difficulties that should be borne in mind in adopting these values. These include: the measure of water quality used; the nature of the water quality–fish stocks linkage and the reliability of the valuation.

Measurement of water quality

River water quality data were supplied by the Institute of Terrestrial Ecology (ITE). The measure of water quality adopted in this study is known as the O/E ASPT. This is the ratio of observed, against expected, average score per taxa (a taxa being a class of organism), derived from knowledge of the location. Alternative measures of water quality based on other biological or chemical measures were considered and on scientific grounds are also regarded as valid. The O/E ASPT measure is used here because it links water quality directly with invertebrates which, in turn, can be related to fish populations. Whilst the different measures of water quality reflect the fact that there are different priorities when examining aquatic environmental impacts, it is also recognised that there exists considerable disagreement as to how a 'pristine' condition water course should be defined, depending on which measure is adopted. Our adoption of the O/E ASPT measure implies one definition and this is useful in the context of angling valuation but we do not suggest that this measure be used in all contexts where damage costs are estimated.

The water quality–angling linkage

The relationship between water quality and fish stocks is complex. There is, for example, no strictly linear relationship between the numbers of invertebrates (as an indicator of water quality) and the number of fish in a river. Other factors, such as water temperature and the nature of the habitat within the stream, impact on fish stocks. Also, a river may be shallow and slow running and therefore not support fish, yet hold plentiful stocks of other organisms by which the water quality indicators are assessed. The relationship can be summarised thus:

$$F = f(q, h, t, Ox, n)$$

where F represents fish numbers, q is the water quality, h is the nature of the habitat, t is the temperature of the water, Ox is the level of depleted oxygen in the water and n is all other factors.

However, a broadly positive relationship between water quality and fish stocks can be discerned which can be characterised as linear as a first approximation. Evidence for this type of characterisation is implicit in the fact that pollution abatement has been shown to lead to an increase in fish stocks in rivers and lakes[1] though it has also been shown that the effects of pollution abatement may take some time to come to fruition. There is, then, a clear impact of anthropogenic activity on the level of fish in rivers.

We analysed the water quality on a hydrometric area basis, taking the mean of the distributional frequency of water quality for sites within the hydrometric area. These areas relate well to large catchment areas but also include small catchments that flow directly to the sea. Water quality was defined at a regional level, this being the basis on which the population data were most easily calculated.

Valuation of angling

Valuation of angling using WTP studies has only recently been developed. The most comprehensive study for the UK is that prepared for the Foundation for Water Research which values the effects of water quality levels on angling across the entire country (FWR, 1995). The results from this study are therefore used as the basis of the current valuation. The study values the impact of variable water quality on coarse and game fishing. Different levels of resulting angling opportunities are classified as Poor, Moderate and Good. WTP values are estimated for each type of angling on a per trip basis. The water quality levels for each region have therefore been classified in three corresponding categories in order to make the valuation possible. The basic valuation data used are shown in Tables 12.3 and 12.4. The valuations are then interpolated between water quality levels in order to relate the valuation data more closely to the average regional water quality levels. This at least allows us to show some regional variation within a classification system that otherwise hides water quality differences.

Table 12.3 Valuation data for different angling types with varying water quality

WTP Values (Euro per trip)			
	Coarse	Game	Salmon
Good	9.44	26.79	31.95
Moderate	5.93	16.01	14.57
Poor	3.95	15.18	N/A

Source: Foundation for Water Research (1995).

Table 12.4 Number of trips per angler per annum for different types of angling and water quality

	Trips per angler per annum		
	Coarse	Game	Salmon
Good	28.94	19.95	9.4
Moderate	17.27	14.91	11.3
Poor	12.77	17.12	N/A

Source: Foundation for Water Research (1995).

It should be noted that the analysis assumes a direct relationship between the value of the angling trip and the probability of catching a fish. In reality it is clear that other factors, such as quality of the surroundings, ease of access, personal challenge, climate, quality of fish and so on, will all be important to individual anglers to a greater or lesser extent.

The average number of trips taken by each angler to a river angling destination per annum, on a regional basis, is known for existing water quality (OPCS, 1991). The number of trips is known for good existing sites and these are taken as a proxy for the number that would be taken with pristine water. As with the valuation data, the number of trips was also interpolated between the basic classifications. From these estimates, the number of trips lost as the result of a fall in water quality from 'pristine' condition can be identified. The resulting change in the participation rate can then be multiplied by the regional population to give the total number of trips lost in the region as a result of the fall in water quality. This total, multiplied by the value change, gives the total damage cost for each type of angling.

(b) Amenity value

For this preliminary exercise we assume that amenity value is reflected in the housing market. Whilst this market is by no means perfect we think it reasonable in the first instance to examine the values that it reveals for amenity. Very simply, therefore, we have valued the change in property price that can be expected to result from comparing prices of properties with and without a water-frontage. The resulting value can therefore, at best, be interpreted as reflecting the difference between existing water quality levels and a water quality level so low as to negate any utility derived from a waterfront location.

An estimate was made of the effects on house prices of water-frontage where water-frontage is taken to mean that a property is within 50 metres of a stretch of water.[2] The studies that have been reviewed suggest that it is reasonable to assume that the premium on house prices lies within the range 2–15 per cent with a mid value of 10 per cent (Willis and Garrod, 1993; ERM Economics, 1997). Average regional house prices were taken as the base level for house prices in each region, the premiums being calculated relative to these prices. This yields a capital value for property premiums on the waterfront, which is then annualised over 25 years at a discount rate of 6 per cent. The annuity is calculated for 25 years since this is the average tenure in an owner-occupied property; the discount rate is the current UK public discount rate. The resulting values of the premiums were then multiplied by the number of households estimated to be within a 50 metre radius of a river in each region (*Social Trends*, 1997).

12.3.2 Results

(a) Angling

The results of this analysis are contained in Table 12.5. For England and Wales the total value of the current recreational benefit for game angling is estimated at 17.2 million Euro. The corresponding value for the recreational benefit if all rivers were improved to 'good' levels of water quality (and, by implication, good angling opportunities) is 19.6 million Euro. Thus, the resulting damages to game fishing in English and Welsh rivers are estimated at 2.5 million Euro.

The values generated for Scotland in terms of salmon angling showed that its recreational benefit currently is 18.7 million Euro, whereas the possible benefit if the water quality were pristine is 29.9 million Euro. Damages were thus estimated to be of the order of 11.3 million Euro. Data on the number of Scottish salmon anglers were provided by the ACA Scotland Anglers' Clearwater Association.[3] No data were available on the number of

Table 12.5 Effect of water quality on angling recreational values (million Euro)

Angling type	Total current value	Total value if good	Total damages
Coarse	18.4	32.5	14.1
Game	35.8	49.6	13.7
Total UK	54.3	82.1	27.8

Source: Foundation for Water Research (1995).

trout anglers, though it was thought that there was considerable overlap between salmon and trout angling.

The total value for the UK in terms of the damages accruing from poor water quality and its impact on game fishing is estimated to be 13.7 million Euro. It must be noted that this is probably an underestimate of the full impact on the game fishing sector as the values for England and Wales used the willingness-to-pay calculations for trout, which are considerably lower than those for salmon.

For coarse fishing, the total recreational benefit accruing from rivers currently is valued at 18.3 million Euro per annum in England and Wales. The corresponding figure if the average water quality improved from its current state to the good level was estimated at 32.3 million Euro. Hence damages of 14 million Euro are estimated. For Scotland, based on data provided by the ACA Scotland Clearwater Anglers' Association, the value of coarse fishing is very low. This is because of the low proportion of Scottish coarse anglers. The value of damages for Scotland was only 0.06 million Euro.

Hence for the UK, total damage to coarse angling amounted to 14.1 million Euro. Thus, the total damage to UK angling from poor water quality (and, by assumption, poor fish stocks) was valued at 27.8 million Euro (June 1997 prices). Table 12.5 displays a summary of these impacts.

(b) Amenity

Table 12.6 provides a summary of the results for the amenity value obtained in this analysis.

From the estimation of the amenity value of waterfront properties it was found that the annualised value was between 3.2 million Euro and 24.2 million Euro,[4] depending on the premium applied to the house price. This suggests that the effect on house prices of water frontage is quite substantial. It must be noted, however, that the values given here are not an indicator of the level of price premium applied to houses by water quality; rather they simply indicate the presence of water. This may overestimate the value in that waterfront properties may be more sparsely distributed than

Table 12.6 Amenity value of riverfront residential property in the UK

Amenity value	Proportion added to value		
	2%	10%	15%
UK total amenity of waterfront (Euro million)	3.2	16.1	24.2

Source: Foundation for Water Research (1995).

those properties further away from the river. It may also underestimate the value in that the waterfront properties are likely to be of a higher standard than the average and therefore the price premium should be applied to a higher than average price.

12.4 CONCLUSIONS

The valuation of the damage costs associated with changes in water quality is very much in its infancy. In section 12.2 a methodology is presented that is basically similar to the restoration costs approach. In fact, water treatment costs can be considered as both the cost that society is willing to pay for making water available for different uses, and the cost society bears for restoring resource quality. In addition, this methodology allows us to identify the economic sectors where the polluting activities originate and eventually to attribute damages and corresponding costs to each economic sector. The main problems for its applicability seem to be the lack of consistent data across countries and of accounting procedures specially for water treatment expenditure.

The study presented in section 12.3 provides a preliminary indication of the damage costs resulting from anthropogenic activity for angling and amenity uses. Clearly, much more work yet remains before this environmental impact is included in environmental accounts on a regular and established basis.

Three points should be noted in conclusion. First, with regard to the valuation of services of water in totality, many other services must be studied including other recreational activities, domestic, agricultural and commercial uses of water and waterways, ecosystems and so on. Both inland waterways and coastal and sea areas need to be covered by a full valuation exercise. The results presented here for angling and amenity values applying to rivers are therefore likely to reflect only a small fraction of annual total damage costs to the water environment.

Second, with regard to the valuation of angling, the complex issue of the interaction between water quality, habitat and angling opportunity needs further study. The appropriate form of scientific measurement of water quality needs to be agreed upon and this then needs to be related to the use of a valuation technique that can adequately assess changes in water quality. The calculations outlined above necessarily make a number of critical assumptions on these issues. We think that these need to be analysed further before these results can be included in environmental accounts with a degree of confidence.

The impact of water quality on house prices and its interpretation as amenity value is contentious. The analysis here only measures the difference between the presence of water (implicitly at its current level of water quality), and its absence. Whilst we have not done so here, it may be possible to derive more meaningful values by comparing house values along stretches of rivers with different water quality levels in order to derive incremental amenity values directly related to water quality.

NOTES

1. See Dauba et al. (1997).
2. Personal communication, Phillip Furze, property valuation expert.
3. Data supplied by Bill Currie, ACA Scotland Anglers' Clearwater Association Chairman, personal communication.
4. Based on a 25-year period and a 6 per cent discount rate.

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Appendix 12.1 Effect of Water Quality on Angling Recreational Values: Detailed Results

Game fishing

Region	Water quality average	Water quality indicator	WTP current	Total value current	Total value if good	Damages (Euro)
Anglian	2.77	1.66	15.73	1 297 593.68	1 539 980.40	2 42 386.72
Northumbria and Yorkshire	3.36	2.02	16.23	2 150 176.88	2 473 301.85	323 124.97
North West	3.22	1.93	15.95	2 273 415.94	2 659 966.14	386 550.19
Severn Trent	2.99	1.79	15.84	4 355 155.28	5 133 267.99	778 112.70
Southern	2.86	1.72	15.78	276 123.65	326 662.51	50 538.86
South Western	3.34	2.00	16.01	1 000 764.38	1 166 651.82	165 887.44
Thames	2.44	1.46	15.56	1 167 109.07	1 399 982.18	232 873.11
Welsh	3.57	2.14	17.52	4 643 030.55	4 946 603.70	303 573.15
England and Wales			16.01	17 163 369.42	196 46 416.57	2 483 047.15
Scotland	3.60	2.16	16.53	18 673 507.19	29 935 803.89	11 262 296.70
UK total		2	16.01	35 836 876.61	49 582 220.45	13 745 343.85

Appendix 12.1 (continued)

Coarse fishing

Region	Water quality average	Water quality indicator	WTP current	Total value current	Total value if good	Damages (Euro)
Anglian	2.77	1.66	5.26	1 301 375	2 535 448	1 234 073
Northumbria and Yorkshire	3.36	2.02	6.00	2 386 638	4 072 083	1 685 446
North West	3.22	1.93	5.79	2 477 294	4 379 410	1 902 116
Severn Trent	2.99	1.79	5.52	4 551 129	8 451 494	3 900 365
Southern	2.86	1.72	5.38	282 311	537 822	255 511
South Western	3.34	2.00	5.93	1 112 698	1 920 794	808 096
Thames	2.44	1.46	4.86	1 093 608	2 304 953	1 211 345
Welsh	3.57	2.14	6.42	5 107 748	8 144 167	3 036 419
England and Wales				18 312 801	32 346 172	14 033 372
Scotland	3.60	2.16	6.49	112 168	176 918	64 750
UK total		2.00	5.93	18 424 969	32 523 090	14 098 122

Source: Foundation for Water Research (1995).

13. Damages to land

Paul Watkiss, Mike Holland, Katie King and Alistair Hunt

13.1 INTRODUCTION

Contaminated land has emerged as a major issue of concern over the past few decades, largely in response to a number of documented problem sites. Of these, the most well known is probably Love Canal in the US, where a housing development was built on a capped disposal site containing hazardous material. Contamination from the site was linked to serious illnesses in the local area and a national emergency was declared, which led in turn to the creation of the Superfund. In Europe, a similar event occurred at Lekkerkerk in the Netherlands, where 1600 drums of illegally dumped toxic waste were discovered. Several hundred houses had been built on the site, which was a reclaimed waste tip, and there were widespread illnesses resulting from contamination of groundwater and underfloor voids.

Such major cases are now extremely rare. Nevertheless, there is still a large legacy of contaminated sites across Europe and whilst waste management practices have improved some land contamination continues.

The assessment of the environmental degradation of land from human activities was not included in the first phase of this research (Markandya and Pavan, 1999). The objective of the following phase of work has therefore been to develop a usable approach to value these damages for the purpose of environmental accounting. This chapter discusses the progress made in assessing damage from contaminated land, looking at:

- impact pathways for contaminated land;
- available databases of contaminated land and evaluation of the application of the available data;
- quantification of expenditure on land remediation.

The analysis is presented in detail for the UK, with additional consideration of data availability and cost estimates for the Netherlands, Germany and Italy.

13.2 CONTAMINATED LAND – IMPACT PATHWAYS

There are a number of possible impact pathways by which contaminated land may damage the environment. The main impacts which can result are:

- effects on human health;
- effects on agriculture and natural ecosystems;
- pollution of water supplies;
- damage to materials;
- public/occupational accidents;
- amenity impacts.

These impacts are discussed in turn below; all are essentially local, but due to pollutant migration, may extend well beyond the boundaries of the site itself.

13.2.1 Human Health Effects

Direct and indirect impact pathways exist for human health effects. The exposure pathways are similar to heavy metals and trace pollutants and were detailed within the recent ExternE Core Project (EC, 1997). In summary, they include:

- direct exposure by inhalation (including soil vapour and dust);
- indirect exposure through dermal contact;
- indirect exposure from ingestion of contaminated soil;
- indirect exposure from ingestion of contaminated food;
- indirect exposure from ingestion of contaminated water.

The health endpoints which may result from exposure are extremely varied because of the wide range of contaminants. They range from major health effects to irritation and may include both acute and chronic effects. Nonetheless it is important to stress that they are potentially significant; these can be seen from the recent concern over possible links between landfill sites and health endpoints such as carcinogenesis and birth defects.

13.2.2 Effects on Agriculture and Natural Ecosystems

The same pathways which can affect humans can also impact on other species and so local livestock or crops may be affected. Plant species are particularly at risk as they have a tendency to accumulate certain compounds in their roots, stems or leaves. Many plants take up heavy metals and at high concentrations (particularly for copper, nickel, zinc and boron) these will either kill the plant or restrict its growth (phytotoxicity). The accumulation of persistent compounds in plants can move up the food chain, affecting ecosystems and humans.

13.2.3 Pollution of Water Supplies

One of the major potential impact pathways is the contamination of groundwater following migration of pollutants from a site. Once a contaminant reaches groundwater it is able to migrate rapidly and provides a direct pathway, via abstracted water, to human exposure. Aquifer pollution is typically long lived and can be measured in decades rather than days or weeks. Similar pollution migration or surface run-off can contaminate surface waters with associated impacts on aquatic ecosystems or on recreational activities.

13.2.4 Damage to Materials

A number of pollutants present in contaminated sites will attack materials. Concrete is susceptible to attack from high sulphate concentrations and plastic pipes are susceptible to attack from certain organic contaminants. Corrosion of materials may lead to damage of building structures or release of other pollutants (from corrosion of storage tanks or pipes).

13.2.5 Public/Occupational Accidents

For a number of contaminated site types there is a real risk of fires and explosions which may result in public and occupational accidents. Fires may be propagated both above and below ground, from coal dust, oils or domestic waste. The greatest risk of explosions comes from methane emissions from old landfill sites; these may be at the site itself or in nearby properties following migration and gas build-up.

13.2.6 Amenity Impacts

There are several effects on local amenity from contaminated land. Contaminated sites will impact through odour, visual intrusion and so on and this will result in property blight.

13.2.7 Quantification of Impacts

As the above discussion shows, there are a significant number of impact pathways which can affect a wide range of receptors. The actual effects will depend on the pollutants present, on the site characteristics and on the local receptor distribution. As a result, effects are extremely site-specific and require individual risk assessment to estimate likely impacts.

It is therefore not possible to estimate likely effects from sites at a national level – as a result, the analysis here instead looks at land remediation expenditure as a proxy for environmental damage. The first step in such an analysis is to look at the availability of data on the scale of the land contamination problem.

13.3 DATA AND INFORMATION ON CONTAMINATED LAND

13.3.1 What is Contaminated Land?

In order to identify the scale of the contaminated land problem, it is important to define exactly what is contaminated land. Unfortunately, definitions vary. They include (NRA, 1994):

- land which represents an actual or potential hazard to health or the environment as a result of current or previous use;
- land which contains substances which, when present in sufficient quantities or concentrations, are likely to cause harm, directly or indirectly, to man, the environment or on occasion to other targets;
- land that contains any substance that when present in sufficient concentrations or amounts presents a hazard. The hazard may be associated with the present status of the land, limit the future use of the land or require the land to be specifically treated before particular use.

There is also often confusion between contaminated and derelict land. The latter can be defined as:

- land so damaged by industrial or other development that it is incapable of beneficial use without treatment.

These definitions are important when collating data on derelict/contaminated land; a derelict site is not necessarily contaminated and likewise a site may be contaminated yet still operational.

13.3.2 Classification of Contaminated Land

Most contaminated sites are a result of past industrial activities, for example, gas works, chemical factories, petrol filling stations, closed landfill sites. Because of this, contaminated land is usually classified in terms of the historical use of a site. The types of activities and the contamination sources they give rise to are shown in Table 13.1.

There is however an alternative way to classify sites: in terms of the specific pollutants (or sub-groups of pollutants) which they contain. There are in the order of 100 substances that fall under this category as being significant with respect to contaminated land evaluation and remediation.

There is invariably some overlap between the two approaches – specific industrial uses tend to give rise to specific pollutant contamination. It is therefore possible to make some generalisations about the types of land contamination risk and clean-up costs for specific site categorisations.

13.3.3 Contaminated Land Legislation and Data Availability in the UK

The history of contaminated land policy in the *United Kingdom* is complex. A national register was proposed in the Environment Protection Act 1990. However, the register was dropped in 1993 due to opposition by property developers and banks. It was superseded by the Environment Act of 1995, which introduced a new regulatory system for the identification and remediation of contaminated land. In April 2000 a statutory regime came into force providing an improved system for the identification and remediation for contaminated land.

As a result, at the present time there is no single comprehensive list indicating the size, location and types of sites that are contaminated for the UK as a whole, though this information will be available in the next few years. Nonetheless, there are estimates of the current levels of contaminated land in the UK and these indicate the problem is significant (HCEC, 1996a, 1996b). Indeed, in terms of damages, it is estimated that between 250 to 500 wells/boreholes have been put out of action as a result of groundwater contamination from contaminated land.

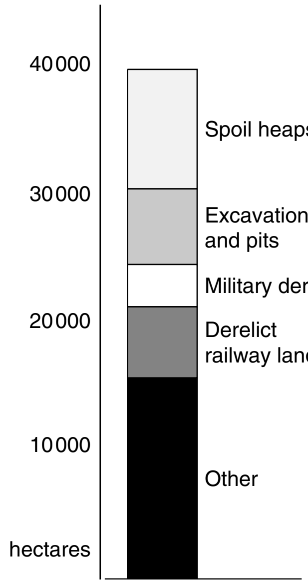
A recent government report on contaminated land and liabilities (DoE, 1993) estimated that there were 39 600 hectares[1] of derelict land within the UK, of which 80 per cent could possibly be contaminated. Of these sites, it is estimated that between 200 and 500 should be classified as 'special' sites, requiring substantial remedial action. Figure 13.1 shows the breakdown from this report by site type.

A number of other estimates exist (for example, Timothy, 1992; MSI, 1996), most of which indicate that the figures above are likely to be a lower

Table 13.1 Activities which may give rise to contamination

Sector	Activities which lead to contamination
Agriculture	Burial of diseased livestock
Extraction	Extraction, handling and storage of fossil fuels
Energy	Producing gas from coal, lignite or other carbonaceous materials
Metals production	Production, refining or recovery of metals
Non-metals production	Production or refining of non-metals by treatment of the ore
Glass making/ceramics	Manufacture of glass/ceramics and products based on glass/ceramics
Production and use of chemicals	Production, refining, recovery or storage of petroleum, petrochemicals or their by-products
Engineering and manufacturing	Manufacture of metal goods. Storage, manufacture or testing of explosives, propellants, ammunition. Manufacture and repair of electrical and electronic components and equipment
Food processing	Manufacture of animal feeds. Processing of animal by-products
Paper, pulp and printing	Production of paper pulp, paper or board, including printing and de-inking
Timber	Chemical treatment and coating of timber/ timber products
Textiles	Processing of natural or synthetic rubber (including tyres)
Infrastructure	Dismantling or maintenance of railway stock, marine vessels, road transport or haulage vehicles
Waste disposal	Treatment of sewage or other effluent. Storage, treatment or disposal of sludge including from water treatment works. Treating, storage or disposing of waste including scrap. Storage or disposal of radioactive material
Miscellaneous	Dry-cleaning operation. Laboratories. Building demolition

Source: NRA (1994).



Source: DoE (1993).

Figure 13.1 Estimated types of derelict land in England

estimate, not least because they exclude land still in use. Applying the estimates from the literature to the total land area of the UK (24 million hectares) would mean between 0.13 and 0.8 per cent of the total land area of the UK being contaminated.

The new regulatory system is based on the 'suitable for use' approach and states that local authorities have a duty to inspect their areas to identify contaminated sites. The inspection strategy relies on desk-based research with Ordnance Survey maps and historical maps to identify past land uses followed by the identification of sites that are most likely to present a risk, that is, sites close to aquifers or residential properties. A site is contaminated if it falls into one of the following proposed categories:

- there is a significant possibility of significant harm being caused;
- there is a significant possibility that pollution of controlled waters is being caused.

The exercise should result in the development of two databases per local authority. The first database will be a compilation of all the suspected contaminated land sites pulled together for the inspection strategy for internal use by the local authority. The second database will follow actual identification of contaminated sites following a site investigation and this information will be used for the register of contaminated land available publicly.

A number of local authorities have already begun to compile this data, for example Wigan (Forster et al., 1995) and St Helens, Merseyside (Doornkamp and Lee, 1992). Such studies provide an indication of the nature and quality of data which might typically be available to form the basis for the required inspection strategy and include information on:

- the number of potentially contaminated sites within the locality;
- the earlier use of the land which has given rise to the potential contamination;
- the spatial extent of the potential contamination.

13.3.4 Contaminated Land Legislation and Data Availability in the Netherlands

The *Netherlands* has probably the most developed legislative programme and most mature contaminated land remediation market in Europe. A number of acts have been implemented since 1982 following the discovery of contaminated soil in a housing development in Lekkerkerk in 1980. The Ministry of the Environment initially ordered the 12 provinces of the Netherlands to compile a list of locations with known or suspected soil contamination, including potential current activities for contamination. This initial investigation identified 4500 sites. This register is known as the Interim Act List and in 1992 held 10 000 sites. The Interim Law on Soil Sanitation (Remediation) of 1983 (Interimwet Bodemsanering) was succeeded by the 1995 Law on Soil Protection (Wet Bodembescherming), which represents the legal framework of soil protection and remediation in the Netherlands.

However, the objectives of soil protection and remediation are currently in flux. Arguably the most important change concerns the necessary degree of remediation of polluted sites. Until recently, the objective was to clean all sites to a degree that would permit ‘multifunctional use’, that is, all sites should be so clean that even the most sensitive activities could be carried out in these sites.

A cabinet decision in 1997 now also permits ‘functional’ remediation, that is, a degree of remediation that takes account of the future use of the site. Long-term objectives of soil remediation policy are that by the year 2010:

- all environmentally urgent sites of soil pollution should be cleaned or protected (isolated);

- the non-urgent sites of serious soil pollution should be protected, if necessary;
- the total size and distribution of soil pollution should be identified.

Soil pollution is considered 'serious' if the intervention values for soil remediation are exceeded, its 'urgency' is determined by local circumstances, for example, the degree of mobility of the pollutants. A complicating factor in the assessment of the number of contaminated sites and the number of remediation projects is that the responsibility for remediation is spread out over a large number of actors (CBS, 1996):

- provinces are responsible for the progress of soil research and remediation in the framework of the Law on Soil Protection;
- the major cities (Amsterdam, Rotterdam, Utrecht and The Hague) have their own soil remediation programmes;
- in 1992 a special programme was started for soil remediation on active industrial sites. Current owners are responsible for research and remediation;
- in 1991 a special remediation programme for petrol stations was started;
- there is a special programme for research and remediation of land of the Dutch railways;
- soil pollution in military areas is under the responsibility of the Ministry of Defence;
- there is a special programme for research and remediation of state-owned land;
- special programmes are aimed at soil research and remediation in the framework of urban development projects.

At this moment there is no uniform national information system on soil pollution and progress of soil remediation. The three western Provinces (North- and South-Holland, and Utrecht) have developed a pilot information system, called the FINABO system. Until recently, there was no registration of remediation projects carried out by private landowners on their own account. RIVM (1997b) concludes that it is not yet possible to give a complete account of the progress of soil remediation in the Netherlands.

In the period 1980–96, the Provinces and the four major cities (Amsterdam, Rotterdam, Utrecht and The Hague) examined 4400 potentially polluted locations and started remediation projects in 2800 locations. A number of these projects have been completed (RIVM, 1997a). There is a lack of knowledge on the number of remediation projects that are carried out by private parties on their own account. Private activities started in the late 1980s. The annual government budget for soil remediation projects is currently about Euro 150 million (RIVM, 1997a). The total estimated number of polluted sites in the Netherlands is 350 000. Of these, 115 000 sites are expected to be seriously polluted, of which 60 000 sites are expected to be classified as 'urgent' (RIVM, 1997b).

Along with soil pollution on land, there is also a serious problem of the pollution of sediments. About 87 million m^3 of sediments are severely polluted. In state waters there are about 200 polluted sites; the number of polluted sites in regional waters is unknown. The remediation of sediments is extremely slow because of a lack of storage capacity for polluted sediments and because of financial reasons (RIVM, 1997b). The annual government budget for sediment remediation projects is currently about Euro 30 million (RIVM, 1997a).

13.3.5 Contaminated Land Legislation and Data Availability in Germany

The current information on contaminated land sites in *Germany*[2] is shown in Table 13.2.

13.3.6 Contaminated Land Legislation and Data Availability in Italy

Contaminated land legislation in *Italy* centres on Law 441 of 29/10/87. This required all regions and the two autonomous provinces (Bolzano and Trento) to set up regional land restoration plans. The criteria and guidelines for making regional restoration plans for contaminated sites were presented in the Ministerial Decree of 16/5/89. Each region has had to set up a list of existing contaminated land and for each area specify the type and source of contamination, the presumed volume of polluted terrain, make a ranking of the sites according to the urgency of restoration action, define short- and medium-term restoration actions for the most urgent situations and estimate restoration costs.

The law has recently been superseded by a Solid Waste Law (Legislative Decree no. 22 of 5/2/97). This is a comprehensive law concerning solid wastes. Article 17 refers to the restoration of polluted sites and sets out the definition of new limits for soil, freshwater and groundwater pollution, new criteria and guidelines for land restoration and definition of plans. At the current time, there has not been much progress, though it is expected that the new guidelines will not be very different from the existing ones. More recently, at the beginning of 1999 a new law has been approved for the financing of remediation activities by the central government.

At the end of 1995, ten regions and the Bolzano province had approved restoration plans in place (approved by the Ministry of the Environment).

Table 13.2 Number of registered sites for which contamination is suspected in Germany

State	Number of registered contaminated sites in Germany		
	Sites polluted through handling of waste	Sites polluted by environmentally damaging materials	Total sites
Baden-Württemberg	5008	1886	6894
Bayern	9549	3029	12578
Berlin	615	5068	5683
Brandenburg	6410	8932	15342
Bremen	100	3000	3100
Hamburg	446	1080	1526
Hessen	145	347	492
Mecklenburg-Vorp.	2810	5890	8700
Niedersachsen	8656	n.a.	8656
Nordrhein-Westfalen	16689	11640	28329
Rheinland-Pfalz	10578	n.a.	10578
Saarland	1801	2442	4243
Sachsen	9211	21120	30331
Sachsen-Anhalt	6742	12716	19458
Schleswig-Holstein	3069	14177	17246
Thüringen	6226	12003	18229
Total	88055	103330	191385

Source: Compiled by the Federal Environmental Agency, Berlin, November 1997.

Other regions had regional approval and a number still did not have a plan. The existing plans are available at the Ministry of the Environment in Rome, or through the different regional offices.

Table 13.3 shows the current situation according to the Italian Report on the State of the Environment (1997). As can be seen, the results are incomplete. Moreover, according to the report, the number of contaminated sites and the estimates of restoration costs appear to be inadequate relative to the regions' size, population and industrial activity. Reported results are probably underestimated for different reasons:

- different criteria have been used by the regions for the definition of contaminated land:
- many of them have not considered active or abandoned industrial sites where polluting activities are continuing, or landfills managed without following legal prescriptions.

Table 13.3 State of land restoration plans (application of Ministerial Decree of 16/5/89)

Region/province	Status	Approval			Short-term cost		Medium-term cost	
		Regional	Ministerial	Sites	No. sites	10^6 Liter	No. sites	10^6 Liter
Piemonte	Completed	3	3	311	26	124 443	266	5 238
Valle d'Aosta	To be done							
Lombardia	Completed	3	3	2 120	19	98 705	70	288 600
Bolzano	Completed	3	3					
Trento	Completed	3						
Veneto	To be done							
Friuli venezia g.	Completed	3	3	151	10	23 570	139	15 183
Liguria	Completed	3	3	85	2	50 354	5	2 795
Emilia romagna	Completed	3	3	3 182	27	85 361	75	67 160
Toscana	Completed	3	3	428	37	65 962	105	3 150
Umbria	Completed	3	3	112	2	13 243	3	622
Marche	Completed			210	10	81 167	131	1 136
Lazio	To be done							
Abruzzo	Completed	3	3	120	4	12 586	8	12 678
Molise	Completed	3	3	30	12	20 186	17	7 597
Campania	To be done							
Puglia	Completed	3		1 212	12	20 576	11	25 555
Basilicata	First phase			411				
Calabria	To be done							
Sicilia	Completed			110	4	14 605	9	
Sardegna	Completed		3	391	79	22 527		
TOTAL				8 873	244	633 285	839	429 715

Source: Ministero dell' Ambiente (1997).

As a result, the number of contaminated sites is probably higher than reported. In addition, different criteria have been used in the calculation of restoration costs: many regions have only considered the costs of surveys and analysis, or the costs of making the sites safe, without achieving a complete restoration.

13.3.7 Contaminated Land Data Availability across the EU

At the *European level* there is no Directive giving guidelines for contaminated land inventories or remediation. The Regulation of the European Communities no. 2242 of 23/7/87 promotes the financing of demonstrative projects aimed at developing techniques for finding and restoring sites contaminated by dangerous substances and wastes. The Resolution of the European Commission of 24/2/97, at point 35, asks member states to adopt measures to guarantee the restoration of contaminated sites and old landfills.

Some information on the status of contaminated land registers within the EU member states can be obtained from a survey carried out in 1993 by the European Commission to establish the development of registers (Bardos et al., 1993). This survey contains the most comprehensive assessment across the EU, though due to the rapidly changing legislation in this area it is now out of date for many countries.

The report concluded that Denmark, the Netherlands and Germany have the most advanced systems for national contaminated land registers. France, Belgium, Ireland, Spain and parts of Italy have undertaken work in compiling registers but it is uncertain how far Greece and Portugal have moved towards initiating this exercise. The contaminated land remediation strategy for different EU member states is summarised in Table 13.4.

Registers are related to the national definition of a contaminated site. In conjunction with a priority setting system, registers are generally regarded as a means of arriving at a consistent, effective and fair allocation of resources for solving contaminated land problems. The consistency in definition varies across the EU and therefore a comparison between registers can only be tentative.

The Caracas Research Programme estimates that there are around 750 000 sites of contaminated land within Europe.

13.4 COSTS OF CONTAMINATED LAND TREATMENT

In order to estimate the environmental damages for land contamination, we next need to look at contaminated land expenditure. The type of

Table 13.4 Contaminated land Remediation Strategy for EU Member States

State	General objectives	Site-specific targets	Approach
Belgium	Reaching an acceptable risk level for public health and the environment	Determined on a site by site basis with regard to future (that is, intended) use	Regional strategies under development. No prescriptive approaches yet but some guidance available in some regions. The Dutch approach may also be followed
Denmark	To deal with contamination so that exposure of humans and environment to hazardous substances is minimised in the long term. That the best possible environmental benefit is achieved for minimum cost	Determined on a site by site basis with regard to current and planned use. Deregistration requires that the site is suitable for the most sensitive use	Nationally developed strategy under constant revision by national and local authorities. Generic guidelines[1] set for most sensitive uses, and for some specific remediation processes
France	Reduce risks to public health and the environment to acceptable levels	Determined on a site by site basis taking into account present and future use of the site	General approach under development. No prescriptive requirements. The Dutch approach is followed in some cases
Germany	Reduce risks to public health and the environment to acceptable levels	Determined on a site by site basis with regard to current and planned use	A variety of regional approaches, with a federal approach under development. Generic guidelines[1] set in many German Länder
Greece	Not specified as regards potential risks. Landscape restoration in some cases	Determined on a site by site basis, main focus often landscape restoration	Ad hoc approach determined on a site by site basis. No nationally set guidance
Ireland	To deal with 'nuisance' and facilitate development	Determined on a site by site basis	Local authorities may assess remediation strategies proposed for individual sites. Some guidance in building regulations
Italy	Where guidance is available (that is, in the Northern	Determined on a site by site basis	Fragmented regional approach. National guidelines may be set in

Table 13.4 (continued)

State	General objectives	Site-specific targets	Approach
	Regions), the general objective is to mitigate the impact of contamination on human health and the environment		the future after submission of regional recommendations. Guidance, including generic approaches[1] developed in some (Northern) regions
The Netherlands	Achieving negligible risk to human health and the environment, within time and budget constraints	As a first preference achieving suitability for any use and full recovery of soil functions (multi-functionality). On a site-specific basis achieving suitability for specific use may be accepted providing contamination is isolated, controlled and monitored	Nationally agreed approach implemented by provincial authorities. Remediation is geared towards restoring soil quality to target levels[1,2] unless proven to be unfeasible. Treatment of soils preferred but disposal to landfill accepted if treatment judged uneconomic by authority
Portugal	Reduce risks to acceptable levels	Determined on an ad hoc site by site basis	Land remediation is considered of secondary importance to the development of waste management policy. However, some national guidance is under development
Spain	Priority to sites where groundwater used as drinking water is polluted and areas which are intended to be reused	Necessity to develop specific legislation	The autonomy of Regions has to be respected
United Kingdom	Ensuring 'suitability for use', that is, ensuring that the quality of land is maintained at a level which enables the continuing best use of land for its current or intended purpose. Cost effectiveness of	Determined on a site by site basis with regard to current and planned use	General approaches must be acceptable to local authority planning control departments and the National Rivers Authority (where groundwater protection is required). Generally arises as part of redevelopment

Table 13.4 (continued)

State	General objectives	Site-specific targets	Approach
	remediation is important. Ensuring that land contamination does not give rise to unacceptable risks to health or safety		

Notes:
[1] Including generic soil and/or groundwater concentration limits for various hazardous substances.
[2] Soil concentration limits are dependent on soil type.

remediation which is used and the remediation costs vary according to a number of parameters, which include:

- location and geological nature of the site;
- type of contaminants;
- soil structure;
- groundwater conditions;
- perceived risks and hazards;
- eventual end-use;
- who is commissioning the project.

The last two points are important as they will vary between countries. The legislation in place will dictate the level of clean-up, that is, whether a multifunctionality or suitable for use approach is taken to remediation. Multifunctionality dictates that sites are remediated to pristine conditions regardless of the end-use for the site. The alternative is the end-use (or suitable for use) approach, which sets the level of remediation according to the end-use of the site, for example, there will be different limits set for a car park as opposed to a housing development. Similarly, there are different ways in which site remediation targets are set. These may mean there are prescriptive levels for individual contaminants or classes of contaminants. The alternative is to take a risk based approach, looking at the contaminant source, the possible impact pathways and the receptor distribution.

The project financiers are also important. The drivers in many countries have been the identification and remediation of sites, irrespective of whether they are to be redeveloped. In the UK, much remediation has been based on commercial worth, that is, market forces determine the redevelopment of potentially contaminated sites with respect to the value of the land after clean-up. In such cases, the degree of remediation is likely to be lower than with government funded projects.

As a result, it is likely that remediation costs will be country specific. Nonetheless, the costs of different technological options should be similar and some data are available on these. There are a number of alternative approaches which can be used to remediate sites, which include:

- leaving the contaminated material in place untreated and monitoring to make sure that contamination does not spread (natural attenuation);
- containment of materials in place, for example by capping the site with concrete;
- removal of contaminants for disposal to landfill/incineration;
- removal of contaminants for ex-situ treatment;
- in-situ treatment of contaminants at the site.

Data on the costs in the UK of these different treatment options are shown in Table 13.5. The data in Table 13.5 could, in theory, be used to estimate likely costs for different site types, based on the most appropriate remediation technology for a particular site type. However, there is a lack of such data in most registers and so at present this is not possible. However, an alternative approach can be used with typical estimates of remediation costs per site:

- *Costs of site investigations* are typically in the range of Euro 15 000– 60 000 per hectare (Royal Commission, 1996; Timothy, 1992).
- *Costs of remediation* are typically in the range Euro 150 000–570 000 (Harris, 1997; Timothy, 1992), with the exception of special sites.

Therefore, the typical costs for remediation are likely to be in the range Euro 167 000–632 000 per hectare, with a mid-estimate of around Euro 380 500 per hectare. These values can be combined with estimates of the total contaminated land area in the UK to estimate total remediation costs (see Table 13.6).

The best estimate of the costs to clean up all contaminated sites in the UK is therefore around 10.4 billion Euro (£8 billion). Given that the current remediation market in the UK is around Euro 915 million/year (MSI, 1996), this represents a very significant remediation liability. Moreover, this figure excludes the worst sites, that is, those with particularly bad contamination. The clean-up costs for such sites can be very much larger and warrant further investigation.

Table 13.5 UK technology costs for different remediation options

Technology	Cost (Euro) per m^3 or per tonne			
	Swannell (1998)	Tillotson (1992)	MSI (1996)	RC (1996)
Containment/removal				
Removal/capping				up to 152/t
Excavation and landfilling	46–122/m^3	53–114/m^3		
Off site incineration	152–609/m^3	152–609/m^3	1141–1522/t	
Physical				up to 928/t
Vacuum extraction		15–99/m^3	61–76/t	
Biological				
Biological treatment		30–91/t		up to 228/t
Bioremediation	15–122/m^3		53–76/t	
Chemical				
Chemical stabilisation	15–152/m^3			
Ex-situ soil washing	23–91/m^3		46–53/tonne	
Thermal				
Thermal treatment				up to 1826/t
Low temp. thermal stripping		76–183 /m^3		
Solidification		15–76/m^3		up to 761/t
Encapsulation	30–274/m^3			
In-situ vitrification		228–457/t	228–381/t	
Vitrification (exc. transport)			61/t	

Note: Exchange rate 26.8.1999: £1 = 0.657 Euro.

Table 13.6 Estimated total costs of land remediation in the UK

Area of contamination	Total remediation costs (billion Euro)		
	low	mid	high
Mid = 32 000 hectares	4.6	10.4	17.3
High = 100 000 hectares	14.3	32.5	54.0

13.4.1 Estimating the Clean-up Costs of Special Sites

To date, there is relatively little data on the restoration costs of major sites within Europe. Some data are available from sites in the UK, for former sites of gasworks, which indicate clean-up costs are of the order of Euro 2.3 million per site. However, it is not known if these are representative of special sites as a whole. Data are, however, available from the US where the remediation market is more mature and where a large number of heavily contaminated sites have been remediated.

The US EPA originally compiled a database of contaminated sites 'Comprehensive Environmental Response, Compensation and Liability Information System' (CERCLIS) following the CERC-Liability Information Act (CERCLA), known as the 'Superfund'.

The programme began as a crash clean-up effort with a trust of \$1.6 billion over a five-year period and increased to \$8.5 billion in 1986. CERCLIS contains over 32 000 sites of contaminated land within the US (Colglazier et al., 1991). Following preliminary assessments, site inspections and a scoring on a hazard ranking system, 1200 sites were proposed for the National Priority List. The number of sites on the NPL is predicted to rise and estimates for resource requirements have been carried out for 1350, 2100 and 3500 sites.

There are good data on the remediation costs of contaminated land for sites on the US's National Priority List (Russel et al., 1996). The information is recorded as Records of Decisions (RODs). Cost information collected from the RODs (for 718 RODs representing 616 sites compiled between 1987 and 1994) includes the overall capital cost, annual operation and maintenance cost and the duration of the treatment. The approach categorised sites into 16 types, as shown in Table 13.7.

The US costs of remediation were calculated on an average basis using cost data from the above study. The most replicable cost data analysis from the research was the 'as-built' cost data as this information takes into account the operating and maintenance costs which are essentially part of the remediation process. This is defined as:

$$\text{As-built cost} = \text{capital cost} + (\text{O\&M} * \text{Duration})$$

where: capital cost = initial construction cost during the first year of remediation; O&M = annual operation and maintenance cost of treatment system over a period greater than one year; duration = length of treatment in years.

A number of adjustments were required to the original values to estimate damages per site. In some cases there was more than one ROD per site.

Table 13.7 Site type and description used in study used in NPL resource requirements

Site type	Contamination description	% of sites
Landfill	Municipal or industrial landfills	27
Surface impoundments	Primarily from stored or hazardous materials in open areas	14
Chemical manufacture	Primarily from the manufacture of chemicals	14
Leaking container	From stored or abandoned containers or hazardous substances	9
Wellfield	From drinking water contamination	7
Metal working	From metal fabrication or recycling operations	6
Wood preserving	From wood treating and preservation processes	
Waste oil	From the handling and recycling of waste oil	
Manufacturing	From the manufacture of other substances	
Asbestos	Asbestos contamination	
Plating	From various electroplating operations	
Drum recycling	From drum recycling operations	
Mining	From mining operations	
Radiological tailings	From the handling of radioactive substances	
TNT processing	From trinotoluene (TNT) processing	
Electrical	From the handling, recycling or manufacturing of electrical components such as transformers, batteries or capacitors	

In addition, the effects of inflation had to be included, plus the cost of essential studies prior to the remediation (estimated at $1 million per site). Furthermore, there was found to be a difference between the predicted costs within the ROD and the actual cost of remediation. The total values per site type, updated for these adjustments, are shown in Table 13.8.

These values can be applied to the equivalent site types in the UK to estimate the stock of special sites. It is estimated that of the contaminated sites

Table 13.8 Remediation costs for contaminated sites

Site type	Cost (m.$)/ROD	Total cost (m.$)*	£ million	million ECU
Landfill	15.17	35.80	23.25	30.29
Surface impoundments	13.13	31.12	20.21	26.33
Chemical manufacture	21.34	49.94	32.43	42.25
Wellfield	14.67	34.66	22.51	29.33
Electrical	9.28	22.29	14.47	18.86
Wood preserving	18.91	44.37	28.81	37.54
Waste oil	12.03	28.6	18.57	24.20
Leaking container	11.66	27.75	18.02	23.48
Manufacturing	12.95	30.71	19.94	25.98
Asbestos	3.76	9.63	6.25	8.15
Plating	7.22	17.55	11.40	14.85
Metal working	13.33	31.56	20.49	26.70
Drum recycling	10.58	25.26	16.40	21.37
Mining	79.07	182.33	118.40	154.27
Radiological tailings	116.14	267.39	173.63	226.24
TNT processing	1.84	5.21	3.38	4.41
Average	22.57	52.76	34.26	44.64

Notes: * With cost adjustment to 1994, assuming 1.48 RODs per site, increasing values by difference between predicted and actual costs (46 per cent) and including $1 million per site for studies prior to remediation. Cost conversions are based on 1994 exchange rates ($1.54:£1:1.30 ECU).

in the UK, up to 500 require substantial action. Statistics submitted to a parliamentary inquiry by DoE in 1989 on the contaminated land problems show the likely categories of these 500 sites.

There is of course the likelihood that definitions of special sites will vary between the US and UK. Nonetheless, a comparison of the percentage of sites classified as requiring special action in the two countries shows a reasonable correlation. In the US, 4 per cent of the 32 000 sites designated for the National Priority List were identified as needing special action. The equivalent figure in the UK is 2.5 per cent of all sites identified. The use of US remediation costs for special sites would therefore seem a reasonable basis as a first approximation of costs.

By comparing site classifications between the US and UK, together with the remediation costs for types of sites, the cost of remediation for

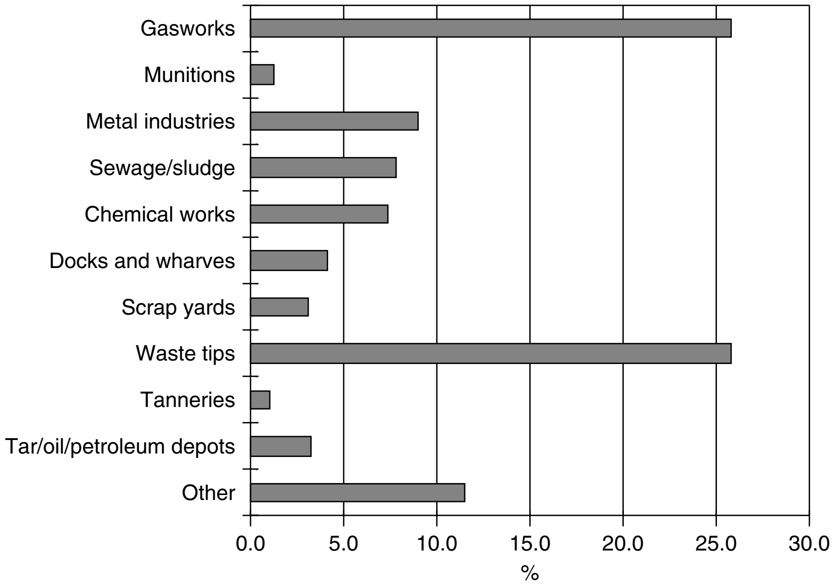


Figure 13.2 Breakdown of 488 estimated 'special' sites in the UK

the UK can be calculated. For some site types, there are obvious site categorisations:

- chemical manufacturing represents 14 per cent of sites in the US and 7 per cent in the UK;
- metal working represents 6 per cent of sites in the US and 9 per cent in the UK;
- landfill represents 27 per cent of sites in the US and 26 per cent (waste tips) in the UK;

Other sites are not obviously comparable to the US categories, therefore, these remaining sites are based on the average remediation cost per site. The results are shown in Table 13.9.

Of course, there are important issues of transferability from the US to the UK, not least the higher costs associated with the Superfund sites, where the technologies were newer (and more expensive) and where government funding was paying for the programme. It would therefore seem that the above method provides an upper estimate of costs of special site remediation in the UK. Based on UK cost data (Euro 2.3 million per site), and extrapolation from US data, the low and high estimates for the UK are shown in Table 13.10.

Table 13.9 Estimated remediation costs for special sites in the UK

Site type (US category)	UK category	No. of UK sites	Cost per site (£ million)	Total (£ million)	Total (Euro million)
Landfill	Waste tips	126	23.25	2929	3817
Chemical manufact.	Chemical works	36	32.43	1167	1521
Metal working	Metal industries	44	20.49	901	1174
Other	Other	282	34.26	9661	12 588
Total		488		14 659	19 101

Table 13.10 Estimated costs of remediation of special sites in the UK

	Total remediation costs (billion Euro)	
No. special sites	Low (UK data)	High (US data)
488	0.95	19.1

The total cost based on the above estimates is therefore between 1–20 billion Euro (£0.75–£15 billion). When combined with the estimate of the bulk of contaminated sites, the total remediation costs for the UK are very large indeed, with a range of 5–75 billion Euro and a best estimate of around 20 billion Euro (£15 billion).

This value can be compared with remediation estimates from other countries studied. Estimates are available for the Netherlands with some preliminary data for Germany. Estimates for land remediation costs in Italy were presented in a previous section.

13.4.2 Estimates of Land Remediation Costs in the Netherlands

The remediation costs of a specific site depend on many factors, including the area of the site, the degree of pollution, the type of pollutants and the type of site (for example, soil characteristics, location and function). In addition, costs are strongly dependent on the remediation target that is chosen. Dellink et al. (1997) have developed a 'cost-effectiveness' curve for soil remediation, in which they estimate the total and annual costs for a complete soil remediation operation at different remediation targets (the 'effect'). They distinguish between '*current targets*', in which not all sites are cleaned to the level of 'multifunctional use'; '*in situ*', in which all sites are cleaned in situ, to

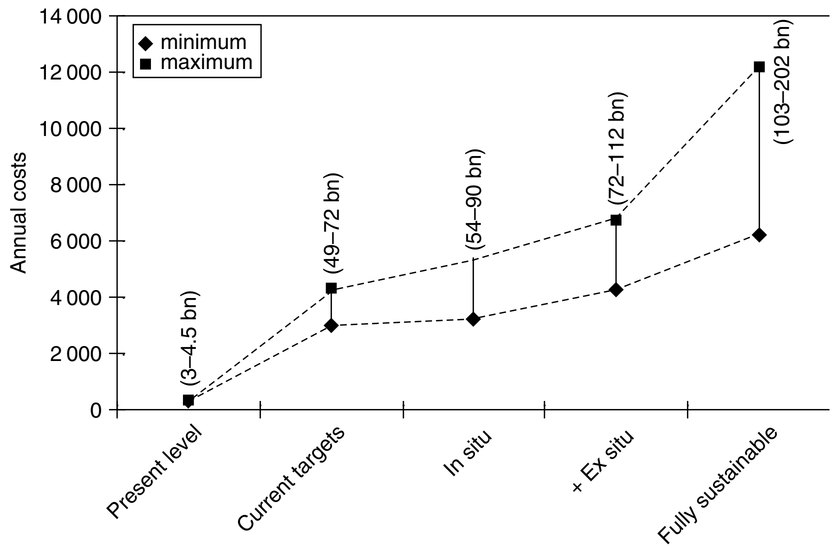


Figure 13.3 'Cost-effectiveness' curve for soil remediation (in billion Euro)

a multifunctional level; '*in and ex situ*', in which the removed earth is also cleaned to a multifunctional level; and finally '*fully sustainable*', in which, in addition to the above operations, a number of large, polluted sites are added whose pollution is currently contained or isolated with technical measures.

The annual total cost of soil remediation is presently around Euro 190 to 270 million. Because of the slow progress of soil remediation in the Netherlands, this figure does not represent the sum that would be needed to achieve the current objectives of the soil remediation policy. The annual cost of soil remediation to meet current targets is estimated to be between Euro 3.0 to 4.3 billion (infinite annuity, interest rate of 6 per cent). With stricter standards, remediation costs could go up to Euro 103 to 202 billion, generating annual costs of Euro 6.2 to 12.1 billion (shown in Figure 13.3).

13.4.3 Estimates of Land Remediation Costs in Germany

Cost data on contaminated sites are not centrally collected. Within the new states, an annual DM 1 billion (about 0.5 billion Euro) is likely to be spent on clean-up between 1992 and 2001. Furthermore for large sites (for example, coal mining sites) in the new states DM 7.5 billion (about 3.85 billion Euro) have been appropriated, of which DM 5.9 billion (about 3.03 billion Euro) had already been spent by 1996 (Sprenger et al., 1997).

Table 13.11 Expenditures (in million Euro) for clean-up of contaminated sites in Bavaria and Hessia (where the polluter is not known)

State	1995	1996	1997
Bavaria*	1.9	4.8	1.5
Hessia**	118.8***	12.7	19.8

Notes:
* The figures relate only to the government part of financing.
** The data refer to the total cost of clean-up.
*** Unlike the data for 1996 and 1997, this is not an annual figure but includes an estimate of total costs for various projects.

Sources: GAB (1998b); HIM (1995, 1996 and 1997).

For the old states it is very difficult to obtain similar data. According to information provided by the Federal Environmental Agency, each state has its own funding body for the clean-up of contaminated sites for those cases where there is no responsible party. Costs borne by private households or firms are very difficult to obtain.

The average clean-up cost per site has been estimated at DM 2 million, which corresponds to about 0.5 million Euro (Bayerisches Staatsministerium für Landesentwicklung und Umweltfragen, 1996).

Some detailed information is available. Figures from Bavaria and Hessia show the current situation with respect to actual expenditures. The following figures relate to contaminated sites where the governments of Bavaria and Hessia have taken action to remedy damage. The respective bodies – GAB in Bavaria and HIM in Hessia – are co-financed by the state government and industrial enterprises.

It is estimated that about 80 per cent of the 12 578 registered sites in Bavaria are not contaminated. Of the remaining 20 per cent of cases which must be cleaned up, about 2 per cent are cases where the polluter can no longer be found and the 'society for remediation of contaminated sites in Bavaria (GAB)' takes responsibility (GAB, 1998a). The GAB paid DM 3.0 million (about 1.53 million Euro) in 1997.

13.5 DISCUSSION AND FUTURE RESEARCH AREAS

At present there is wide variation concerning data availability in different EU countries. The fact that there is no standard classification means it is difficult to compare information and costs between countries. There are some European programmes in place: DGXII has established the Concerted Action for Risk Assessment for Contaminated Sites (CARACAS) and the Network for Industrial Land in Europe (NICOLE) programmes. However, it is unlikely that procedures for identification and classification of contaminated land will be standardised across Europe in the near future. Due to the variance between countries in the definition of contaminated land and hence in the format of national registers (where held), some form of comparison methodology would need to be evaluated.

Similarly, the remediation costs in different countries vary widely according to the legislation in place. Improvements in costs of remediation could be achieved by using estimates per site type. This is dependent on the registers including information on past industrial use, or contaminants present. Nevertheless, because of the differences in legislation, such an approach would still need to be country specific.

Given the site specificity of contaminated land and potential effects, it seems unlikely that an impact pathway approach is possible, quantifying effects on human health, crops and so on. Instead the analysis here has looked at remediation costs. However, the assessment within this study has only quantified the stock of contaminated land. What is needed as the basis of environmental accounting is the change in contaminated land from anthropogenic activity on an annual basis (that is, the flow of damage). It is possible to estimate what the remediation expenditure per year would be for a number of the countries studied, say within a given time frame (for example, by assuming the target for remediation of all current sites is 2010). However, these remediation costs will be for remediating land contaminated as a result of historical contamination, not current contamination. Therefore, further work is needed in looking at the levels of current day land contamination, which, though lower, do continue through activities and accidental releases.

There are also important issues of cost internalisation for contaminated land, as to whether lower values of land are already reflected in the present accounts. It has not been possible to look at this issue within the current study, though this is highlighted as an area requiring further research in the future.

There are a number of difficult issues that need to be addressed to bring analysis of contaminated land to a standard similar to that of the assessment of air pollution. However, data on numbers of sites, site area and current annual clean-up costs are available. These data suggest that annual change in the stock of contaminated land could now be estimated in some regions. This is an important step towards a more complete integration of effects within an accounting framework.

13.6 CONCLUSIONS

This study is an initial attempt to quantify the value of contamination damages for the purpose of environmental accounting. This chapter has reviewed the legislation and availability of data on contaminated land in the UK, the Netherlands, Germany and Italy. The review has shown that data availability and quality vary significantly from country to country and this has meant that it has not been possible to adopt a uniform approach. Similarly, remediation costs vary widely in different countries because of the legislation in place affecting end-use and technology options. As a result, it is likely that any approach to classifying and quantifying remediation costs will need to be country specific.

The information on data availability and remediation expenditure and liabilities is summarised in Table 13.12. This shows that it is likely that there are wide differences in site registration and remediation data between countries and the numbers cannot be compared directly.

Generic cost estimates are available for site remediation costs. In addition, within all countries there are likely to be a small percentage of sites which require significant remediation action and have much higher costs. There are good data from the US on the costs of such treatment, though there are important questions of transferability regarding application in Europe, and the US data can only be used as an upper estimate.

Table 13.12 Summary of contaminated land sites and costs

Country	No. of sites	No. of 'special sites'	Current remediation expenditure (Euro per yr)	Total remediation liability (Euro)
UK	20 000 [1]	488	780 million ECU	30 billion
Italy	8 873	n.a.	n.a.	0.55 billion [2]
Netherlands	350 000 [3]	60 000 [4]	190–270 million	103–202 billion [5]
Germany	191 385 [6]	n.a.	n.a.	35–178 billion [6]

Notes:
[1] Upper estimate (not shown) is 100 000 sites.
[2] Based on remediation costs for short- and medium-term sites identified.
[3] Of which 115 000 seriously polluted.
[4] Classified as 'urgent'.
[5] With stricter standards in place.
[6] Not all registered sites are contaminated. Remediation liability quantified using costs of DM 2 million per site. Low estimate assumes only 20 per cent of sites are contaminated.

In summary, the assessment within this study has made a first approximation of the stock of contaminated land. What is needed for the basis of environmental accounting is the change in contaminated land from anthropogenic activity on an annual basis (that is, the flow of damage). Although it is possible to estimate these numbers based on the total remediation liability, these costs will be for the remediation of land contaminated as a result of historical activity, not current contamination. Therefore, further work is needed in looking at the levels of current day land contamination.

Finally, the study has identified a number of other areas which require future research:

- comparison of the definitions and data quality of contaminated land registers across Europe to allow evaluation of sites and costs in different countries;
- improvements in costs of remediation by using estimates per site type, based on historical use. These estimates are likely to remain country specific;
- improvements in the costs of remediation for special sites, with the use of European rather than US data;
- assessment of possible cost internalisation issues for contaminated land.

NOTES

1. 2.47 acres = 1 hectare; 1 km^2 = 100 hectares.
2. The data reflect the state of registration of sites which are suspected to be contaminated in each federal state. However, due to differences in definitions in each state, the figures are not comparable between the states (for example, data for Hessen shows the number of sites for which contamination is proven).

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14. Marginal cost estimates of greenhouse gas emissions*

Richard S.J. Tol, Samuel Fankhauser, Onno Kuik and Joel B. Smith

14.1 INTRODUCTION

Many economic activities generate waste in the form of greenhouse gases that are emitted into the atmosphere. The build-up of greenhouse gases in the atmosphere is believed to affect the world's future climate. Climate change continues to figure prominently as one of the major environmental concerns for the future. Some people argue climate change is a problem because it could cause unacceptable hardship for particularly vulnerable populations (for example, those living on small island states). Others are concerned about the potential threat to certain unique and valuable systems (such as coral reefs). Still others worry that climate change will increase the probability of large-scale climate instabilities (for example, a shutdown of the Gulf Stream), and will have costly impacts on economies through floods and storms. A fourth group wonders about the total (or aggregate) impacts of climate change. They argue that emission reduction is costly too, that balancing abatement costs against the damages avoided from climate change is a way of determining what action should be taken (cf. Smith et al., 2001). The subject of this chapter – the marginal costs of climate change – particularly addresses the need for and problems in calculating aggregate costs of climate change.

Economic activities that emit greenhouse gases generate intertemporal externalities. The future damage of current emissions should be accounted for in a system of green accounts. A key challenge when assessing the impacts of climate change is synthesis, that is, the need to reduce the complex pattern of local and individual impacts to a more tractable set of indicators. The challenge is to identify indicators that can summarise and make comparable the impacts in different regions, sectors or systems in a meaningful way. Various 'physical' indicators have been advanced, such as number of people affected (for example, Hoozemans et al., 1993), change in total plant growth (White et al., 1998), run-off (Arnell, 1999) and number of systems undergoing change (for example, Alcamo et al., 1995). Such physical metrics suffer from being inadequately linked to human welfare. If the aim is to integrate the impacts of climate change with standard national accounts, it is necessary to express the costs of climate change in the same metric, that is, money (for estimates, see Ayres and Walter, 1991; Cline, 1992; Downing et al., 1995, 1996; Hohmeyer and Gaertner, 1992; Fankhauser, 1995; Nordhaus, 1991, 1994; Mendelsohn and Neumann, 1999; Titus, 1992; Tol, 1995). Money is a particularly well suited metric to measure market impacts, that is, impacts that are linked to market transactions and directly affect GDP (that is, a country's national accounts). The costs of sea level rise can be expressed as the capital cost of protection plus the economic value of land and structures at loss or at risk; and agricultural impact can be expressed as costs or benefits to producers and consumers. Using a monetary metric to express non-market impacts, such as effects on ecosystems or human health, is more difficult. There is a broad and established literature on valuation theory and its application, including studies (mostly in a non-climate change context) on the monetary value of lower mortality risk, ecosystems, quality of life and so on (for example, Freeman, 1993). But economic valuation can be controversial, and requires sophisticated analysis that is mostly still lacking in a climate change context (for example, Pearce et al., 1996).

This chapter presents a brief overview of assessments of global warming damages and assessments of the *present value* of future damages of current emissions of greenhouse gases. This chapter draws heavily on the review of literature that was carried out for the Intergovernmental Panel on Climate Change's (IPCC) Second and Third Assessment Reports (Pearce et al., 1996; Smith et al., 2001), the EU-DG Research ExternE and GARP projects (European Commission, 1998; Eyre et al., 1999a,b; Tol and Downing, 2000) and related publications (Tol, 1999; Tol et al., 2000, forthcoming).

The assessment of monetary damages of emissions of greenhouse gases is usually carried out in three steps. First, the impacts that a specific level of climate change would have on the *present* economy is assessed . Usually those climate damages are assessed that are due to a doubling of the concentration of greenhouse gases in the atmosphere relative to their pre-industrial level. The scenario that describes the impacts of a doubling of greenhouse gas concentrations on the present economy is called the $2 \times CO_2$ scenario. The second step assesses how developments in population, economy, technology and socio-political factors affect future climate change damages. The third step assesses the contribution of the present emission of one unit of greenhouse gas to the flow of future damages.

The discounted sum of these marginal future damages measures the external damage costs of the present emission of one unit of greenhouse gas.

14.2 EQUILIBRIUM 2 × CO_2 IMPACTS

The first step in the damage assessment is the assessment of global warming damage under the 2 × CO_2 scenario, the hypothetical scenario that imposes the impacts of a doubling of the concentration of greenhouse gases in the atmosphere on the present economy. Impacts of this scenario fall both on market goods and on non-market goods, such as human health, biodiversity and ecosystems. Most of the research on the assessment of these impacts is carried out in and for the United States (see Table 14.1).

The selected studies in Table 14.1 report aggregate damage figures that are in a relatively close range (1.0 to 2.5 per cent of GDP). This can be explained from the fact that the studies often use the same sources with respect to the assessment of physical impacts and that the studies make use

Table 14.1 Assessments of global warming damage in the United States (2 × CO_2 scenario, billion US$, various studies)

	Fankhauser (2.5°C)	Cline (2.5°C)	Nordhaus (3.0°C)	Titus (4.0°C)	Tol (2.5°C)
Coastal defence	0.2	1.0	7.5	–	0.2
Dry land	2.1	1.5	3.2	–	1.4
Wetland	5.6	3.6	–	5.0	12.4
Species	7.4	3.5	–	–	20.8
Agriculture	0.6	15.2	1.0	1.0	10.6
Forestry	1.0	2.9	–	38.0	–
Energy	6.9	9.0	–	7.1	–
Water	13.7	6.1	–	9.9	–
Other sectors	–	1.5	38.1	–	–
Mortality/ morbidity	10.0	>5	–	8.2	8.6
Air pollution	6.4	>3.0	–	23.7	–
Water pollution	–	–	–	28.4	–
Migration	0.5	0.4	–	–	0.6
Natural hazards	0.2	0.7	–	–	0.6
Total USA	60.2	>53.3	50.3	121.3	68.4
(% GDP)	(1.2)	(>1.1)	(1.0)	(2.5)	(1.3)

Source: Pearce et al. (1996).

of each other's results. Note that the figures in Table 14.1 are central estimates or 'best guesses'; below we will also consider the probability distributions around these central estimates. Table 14.1 presents damage estimates for 14 different impact categories. A detailed description of each can be found in Pearce et al. (1996).

A number of studies have estimated the total impact of climate change (disaggregated across sectors) in different regions of the world. Table 14.2 shows aggregate, monetised impact estimates for a doubling of atmospheric carbon dioxide on the current economy and population from the three main studies undertaken since the IPCC Second Assessment Report (Pearce et al., 1996), and summarises the 'first generation' of studies already reviewed in the Second Assessment Report for comparison. The numerical results remain speculative, but they can provide insights on signs, orders of

Table 14.2 Estimates of the regional impacts of climate change[a]

	'First generation'	Mendelsohn et al.		Nordhaus / Boyer	Tol[b]
	2.5°C	1.5°C	2.5°C	2.5°C	1.0°C
North America	−1.5				3.4(1.2)
USA	−1.0 to −1.5		0.3	−0.5	
OECD Europe	−1.3				3.7(2.2)
EU	−1.4			−2.8	
OECD Pacific	−1.4 to −2.8				1.0(1.1)
Japan			−0.1	−0.5	
Eastern Europe and former USSR	0.3				2.0(3.8)
Eastern Europe				−0.7	
Former USSR	−0.7				
Russia			11.1	0.7	
Middle East	−4.1			−2.0[c]	1.1(2.2)
Latin America	−4.3				−0.1(0.6)
Brazil			−1.4		
South and Southeast Asia	−8.6				−1.7(1.1)
India			−2.0	−4.9	
China	−4.7 to −5.2		1.8	−0.2	2.1(5.0)[d]
Africa	−8.7			−3.9	−4.1(2.2)
Developed countries (DCs)		0.12	0.03		
Less developed countries (LDCs)		0.05	−0.17		

Table 14.2 (continued)

	'First generation'	Mendelsohn et al.		Nordhaus / Boyer	Tol[b]
	2.5°C	1.5°C	2.5°C	2.5°C	1.0°C
World					
Output weighted[e]	−1.5 to −2.0		0.1	−1.5	2.3(1.0)
Population weighted[f]				−1.9	
At world average prices[g]					−2.7 (0.8)
Equity weighted[h]					0.2(1.3)

Notes:

[a] Figures are expressed as impacts on a society with today's economic structure, population, laws and so on. Mendelsohn et al.'s estimates denote the impact on a future economy. Estimates are expressed as a percentage of Gross Domestic Product. Positive numbers denote benefits, negative numbers denote costs.

[b] Figures in parentheses denote standard deviations. They denote a lower bound to the real uncertainty.

[c] High-income OPEC.

[d] China, Laos, North Korea, Vietnam.

[e] Regional monetary impact estimates are aggregated to world impacts without weighting.

[f] Regional monetary impact estimates are aggregated to world impacts using weights that reflect differences in population sizes.

[g] Regional impacts are evaluated at world average values and then aggregated, without weighting, to world impacts.

[h] Regional monetary impact estimates are aggregated to world impacts using 'equity weights' that equal the ratio of global average per capita income to regional average per capita income.

Source: Pearce et al. (1996); Mendelsohn et al. (1996); Nordhaus and Boyer (2000); Tol (2002a,b).

magnitude and patterns of vulnerability. Results are difficult to compare because different studies assume different climate scenarios, make different assumptions about adaptation, use different regional disaggregation and include different impacts. The Nordhaus and Boyer (1999) estimates, for example, are more negative than others, partly because they factor in the possibility of catastrophic impact. The Mendelsohn et al. (1996) and Tol (2002a,b) estimates, on the other hand, are driven by optimistic assumptions about adaptive capacity and baseline development trends, which result in mostly beneficial impacts.

Standard deviations are rarely reported, but probably amount to several times the 'best guess'. They are larger for developing countries, where results are generally derived through extrapolation rather than direct estimation.

This is illustrated by the standard deviations that are reported in parentheses and estimated by Tol (2002a,b), as reproduced in Table 14.2. Downing et al. (1996) provide a much higher range of uncertainty, from nearly zero impact to almost 40 per cent of world GDP, reflecting a much wider range of assumptions than are commonly included. The Tol estimates probably still underestimate the true uncertainty, for example because they exclude omitted impacts and severe climate change scenarios.

Overall, the current generation of aggregate estimates may understate the true cost of climate change because they tend to ignore extreme weather events, underestimate the compounding effect of multiple stresses and ignore the costs of transition and learning. However, studies may also have overlooked positive impacts of climate change and not adequately accounted for how development could reduce the impacts of climate change. Our current understanding of (future) adaptive capacity, particularly in developing countries, is too limited, and the inclusion of adaptation in current studies too varied to allow a firm conclusion about the direction of the estimation bias.

While our understanding of aggregate impacts remains limited, it is constantly improving. Some sectors and impacts have gained more analytical attention than others, and as a result are better understood. Agricultural and coastal impacts in particular are now well studied. Knowledge about the health impacts of climate change is also growing. Several attempts have been made to identify other non-market impacts, such as changes in aquatic and terrestrial ecological systems, and ecosystem services, but a clear and compatible quantification has not yet emerged. A few generic patterns and trends are nevertheless appearing.

Market impacts are low, and may be positive in some countries and sectors – at least in developed regions. This is largely due to adaptation. Efficient adaptation reduces the net costs of climate change because the cost of such measures is lower than the concomitant reduction in impacts. However, impact uncertainty and lack of capacity may make efficient and error-free adaptation difficult. Even so, market impacts could be significant in some conditions, such as a rapid increase in extreme events, which might lead to large losses and/or costly over-adaptation (if random fluctuations are mistaken for a trend) (see Downing et al., 1998).

Developing countries are more vulnerable to climate change than developed countries because their economies rely more heavily on climate-sensitive activities (in particular agriculture), and many already operate close to environmental and climatic tolerance levels (for example, with respect to coastal and water resources). Developing countries are poorly prepared to deal with the climate variability and natural hazards they already face today (World Bank, 2000). If current development trends continue, few of them will have the financial, technical and institutional capacity and knowledge base to deal with the additional stress of climate change.

Differences in vulnerability will not only be observed between regions, but also within them. Some individuals, sectors and systems will be less affected, or may even benefit, while other individuals, sectors and systems may suffer significant losses. There are indications that poor people in general, wherever they live, may be more vulnerable to climate change than the better off. Differences in adaptive capacity are again a key reason for this pattern.

Estimates of global impact are sensitive to the way figures are aggregated. Because the most severe impacts are expected in developing countries, the more weight is assigned to developing countries, the more severe are aggregate impacts (see the next section). Using a simple adding of impacts, some studies estimate small net positive impacts at a few degrees of warming, while others estimate small net negative impacts.

Net aggregate benefits do not preclude the possibility of a majority of people being negatively affected, and some population groups severely so. This is due to the fact that developed economies, many of which could have positive impacts, contribute the majority of global production but account for a smaller fraction of world population. However, there are no studies so far that have consistently estimated the total number of people that could be negatively affected by climate change.

The need for synthesis and aggregation poses challenges with respect to the spatial and temporal comparison of impacts. Aggregating impacts requires an understanding of (or assumptions about) the relative importance of impacts in different sectors, in different regions and at different times. Developing this understanding implicitly involves value judgements. The task is simplified if impacts can be expressed in a common metric, but even then aggregation is not possible without value judgements. Aggregation across time, and the issue of discounting, is discussed by Arrow et al. (1996) and Portney and Weyant (1999). Discounting is particularly important for estimating the marginal costs of greenhouse gas emissions, an issue we return to in Section 14.4. The value judgements underlying regional aggregation are discussed and made explicit in Azar (1999), Azar and Sterner (1996) and Fankhauser et al. (1997, 1998). We underline the importance of aggregation by using four alternative ways of computing world total impacts from regional impact estimates in Table 14.2.

The impact estimates in Tables 14.1 and 14.2 are very uncertain, and the studies upon which they are based suffer from many shortcomings. We list the most important.

A major difficulty in impact assessment is our still incomplete understanding of climate change itself, in particular the regional details of climate change (Mahlman, 1997). Impacts are local, and impacts are related to weather variability and extremes. Current climate change scenarios and current climate change impact studies use crude spatial and temporal resolutions, too crude to capture a number of essential details that determine the impacts.

Knowledge gaps continue at the level of impact analysis. Despite a growing number of country-level case studies (for example, US Country Studies Program, 1999), our knowledge of local impacts is still too uneven and incomplete for a careful, detailed comparison across regions. Furthermore, differences in assumptions often make it difficult to compare case studies across countries. Only a few studies try to provide a coherent global picture, based on a uniform set of assumptions. The bases of many such global impact assessments tend to be case studies with a more limited scope, often undertaken in the United States, which are then extrapolated to other regions. Such extrapolation is difficult and will be successful only if regional circumstances are carefully taken into account, including differences in geography, level of development, value systems and adaptive capacity. Not all analyses are equally careful in undertaking this task. While our understanding of the vulnerability of developed countries is improving – at least with respect to market impacts – good information about developing countries remains scarce.

Non-market damages, indirect effects (for example, the effect of changed agricultural output on the food processing industry), horizontal interlinkages (for example, the interplay between water supply and agriculture; or how the loss of ecosystem functions will affect GDP), and the sociopolitical implications of change are also still poorly understood. Uncertainty, transient effects (the impact of a changing rather than a changed and static climate), and the influence of change in climate variability are other factors deserving more attention.

Another key problem is adaptation. Adaptation will entail complex behavioural, technological and institutional adjustments at all levels of society, and not all population groups will be equally adept at adapting. Adaptation is treated differently in different studies, but all approaches either underestimate or overestimate its effectiveness and costs. Impact studies are largely confined to autonomous adaptation, that is, adaptations that occur without explicit policy intervention from the government. But in many cases governments too will embark on adaptation policies to avoid certain impacts of climate change, and may start those policies well before critical climatic change occurs – for example, by linking climate change adaptation to other development and global change actions, such as on drought and desertification or biodiversity.

The analysis is further complicated by the strong link between adaptation and other socio-economic trends. The world will change substantially in the future, and this will affect vulnerability to climate change. For example, a successful effort to roll back malaria could reduce a climate change-induced spread of malaria risks. A less successful effort could introduce antibiotic-resistant parasites or pesticide-resistant mosquitoes, increasing vulnerability to climate change. The growing pressure on natural resources from unsustainable economic development is likely to exacerbate the impacts of climate change. However, if this pressure leads to improved management (for example, water markets), vulnerability might decrease. Even without explicit adaptation, impact assessments therefore vary depending on the 'type' of socio-economic development expected in the future. The sensitivity of estimates to such baseline trends can in some cases be strong enough to reverse the sign, that is, a potentially negative impact can become positive under a suitable development path or vice versa (Mendelsohn and Neumann, 1999).

14.3 FUTURE DEVELOPMENTS

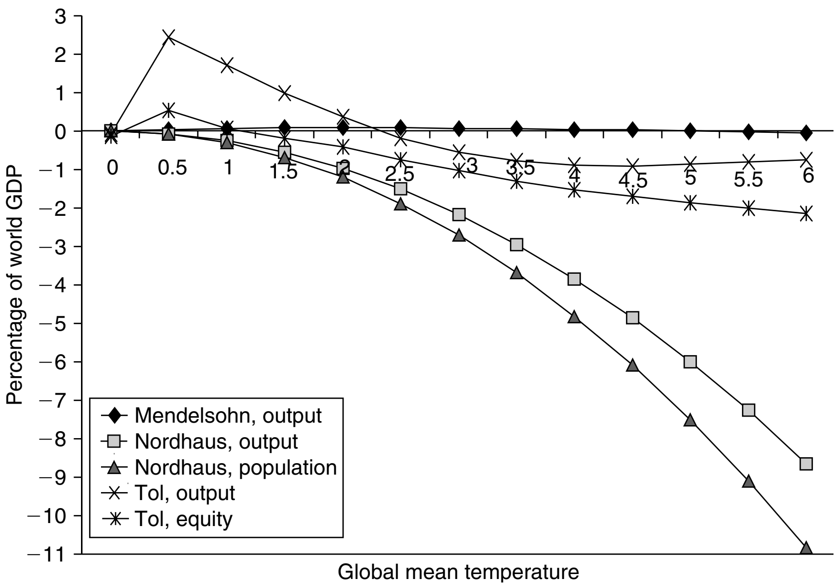
Global warming damages under the $2 \times CO_2$ scenario are hypothetical damages for the *present* society. The real damages of global warming will occur in the future, and society will have changed in terms of population numbers, economic size and structure, technology and in terms of socio-cultural and political factors. These changes may affect the vulnerability of society to global warming. If we are interested in the absolute size of damages (for example, per ton of CO_2 emissions) it is necessary to establish the future size of the population and stock at risk. The timeframe of global warming damages is too long, however, to predict such future developments with any measure of precision. Hence, scenarios are used that describe *possible* futures but do not claim to describe the *most likely* future. Because of the use of these scenarios, global warming damage assessments have a contingent nature: they are contingent upon the scenario assumptions used.

One of the main challenges of impact assessments is to move from this static analysis to a dynamic representation of impacts as a function of shifting climate characteristics, adaptation measures and exogenous trends like economic and population growth. Our understanding of the time path aggregate impacts will follow under different warming and development scenarios is still extremely limited. Among the few explicitly dynamic analyses are Sohngen and Mendelsohn (1999), Tol and Dowlatabadi (2002) and Yohe et al. (1996). These studies are highly speculative, as the underlying models only provide a very rough reflection of real-world complexities. Figure 14.1 shows examples from three studies. While some analysts still work with relatively smooth impact functions (for example, Nordhaus and

Boyer 2000), there is growing recognition (for example, Tol, 1996, 2002a,b; Mendelsohn and Schlesinger, 1999) that the climate impact dynamics – the conjunction of climate change, societal change, impact, and adaptation – is certainly not linear, and might be quite complex.

Impacts in different sectors may unfold along fundamentally different paths. Coastal impacts, for example, are expected to grow continuously over time, more or less in proportion to the rise in sea level. The prospects for agriculture, in contrast, are more diverse. While some models are already predicting aggregate damages for moderate warming, many studies suggest that under some (but not all) scenarios the impact curve might be hump-shaped, with short-term (aggregate) benefits under modest climate change turning into losses under more substantial change (for example, Mendelsohn and Schlesinger, 1999).

Mendelsohn et al. aggregate impacts across different regions weighted by regional output. Nordhaus and Boyer's aggregations are weighted by either regional output or regional population. Tol aggregates either by regional output or by equity, that is, by the ratio of world per capita income to regional per capita income.



Sources: Mendelsohn et al. (1996), Nordhaus and Boyer (2000), and Tol (2002a,b).

Figure 14.1 The impact of climate change as a function of the global mean temperature

14.4 ASSESSMENT OF MARGINAL DAMAGES

The marginal damages caused by a metric ton of carbon dioxide emissions in the near future were estimated by a number of studies, reproduced in Table 14.3. Most estimates are in the range \$5–20/tC (per ton of carbon), but higher estimates cannot be excluded. The uncertainty about marginal damage costs is right-skewed, so the mean is higher than the best guess, and nasty surprises are more likely than pleasant surprises. Several studies confirm that the marginal costs are extremely sensitive to the discount rate.

The alternative estimate of Tol (1999) uses equity weighting, an aggregation procedure that takes into account that a dollar is worth more to a poor person than to a rich one (see Fankhauser et al., 1997, 1998). Equity weighting puts more emphasis on impacts in developing countries, so the marginal damage cost estimate is considerably higher.

Table 14.3 Estimates of the marginal damage costs of carbon dioxide emissions (in \$/tC)

Study/PRTP[a]	0%	1%	3%
Nordhaus (1994)			
• Best guess			5
• Expected value			12
Peck and Teisberg (1992)			10–12
Fankhauser (1995)[b]		20 (6–45)	
Cline (1992, 1993)		6–124	
Plambeck and Hope (1996)[c]	440	46	21
	(390–980)	(20–94)	(10–48)
Tol and	20	4	−7
Downing (2000)[d]	75	46	16
Tol (1999)[e]			
• Best guess	73	23	9
• Equity weighted	171	60	26

Notes:

[a] Pure rate of time preference, or utility discount rate. The more conventional consumption, or money discount rate equals the utility discount rate plus the growth rate of per capita income.

[b] Expected value, uncertainty about the discount rate included.

[c] Plambeck and Hope (1996) use pure rates of time preference of 0, 2 and 3 per cent. The range is the 95 per cent confidence interval (parametric uncertainty only).

[d] Tol and Downing (2000) report estimates from Tol's *FUND* model (top line) and from Downing's *Open Framework* model (bottom line).

[e] Tol uses consumption discount rates of 1, 3 and 5 per cent; the assumed per capita income growth is roughly 2 per cent.

Table 14.4 Estimates of the marginal damage costs of methane emissions (in $/tCH_4$)

Study/PRTP[a]	0%	1%	3%
Fankhauser (1995)[b]		108 (48–205)	
Tol and	−90	−117	−119
Downing (2000)[c]	256	233	139
Tol (1999)[d]			
• Best guess	141	89	52
• Equity weighted	517	295	170

Notes:

[a] Pure rate of time preference, or utility discount rate. The more conventional consumption, or money discount rate equals the utility discount rate plus the growth rate of per capita income.

[b] Expected value, uncertainty about the discount rate included.

[c] Tol and Downing (2000) report estimates from Tol's *FUND* model (top line) and from Downing's *Open Framework* model (bottom line).

[d] Tol uses consumption discount rates of 1, 3 and 5 per cent; the assumed per capita income growth is roughly 2 per cent.

All the studies in Table 14.3 are based on what we have called the 'first generation' of impact studies, except for the Tol estimate in Tol and Downing (2000). This last study uses more optimistic estimates of the impact of climate change (cf. Table 14.2). Consequently, marginal damage costs are low and, for a high discount rate, may even be negative (that is, marginal benefits).

There are fewer estimates of the marginal costs of other greenhouse gases. Table 14.4 displays some estimates for methane (CH_4) and Table 14.5 for nitrous oxide (N_2O).

Uncertainties abound in climate change. The uncertainties about the impact of climate change are estimated in Tol (2002a,b). These are confounded by uncertainties about the scenarios (which expand through time) and about the workings of the climate system. The exact specification can be found in Tol and Downing (2000). The uncertainty analysis is restricted to parametric uncertainty. The uncertainties reflect the ranges found in the literature. Thus, the uncertainty calculated below is a lower bound to the 'true' uncertainty. This particularly holds for the uncertainties about the impacts of climate change. The literature on that subject is thin (excepting agriculture), and ranges are therefore narrow.

Figure 14.2 depicts the uncertainty about the marginal costs of carbon dioxide emissions, based on a Monte Carlo analysis with 1000 runs. The best guess is $9/tC.[1] The uncertainty is large and right-skewed. The probability

Table 14.5 Estimates of the marginal damage costs of nitrous oxide emissions (in $\$/tN_2O$)

Study/PRTP[a]	0%	1%	3%
Fankhauser (1995)[b]		2895 (805–7235)	
Tol and	2351	782	−270
Downing (2000)[c]	11385	6636	2078
Tol (1999)[d]			
• Best guess	7559	2201	817
• Equity weighted	16862	5459	2217

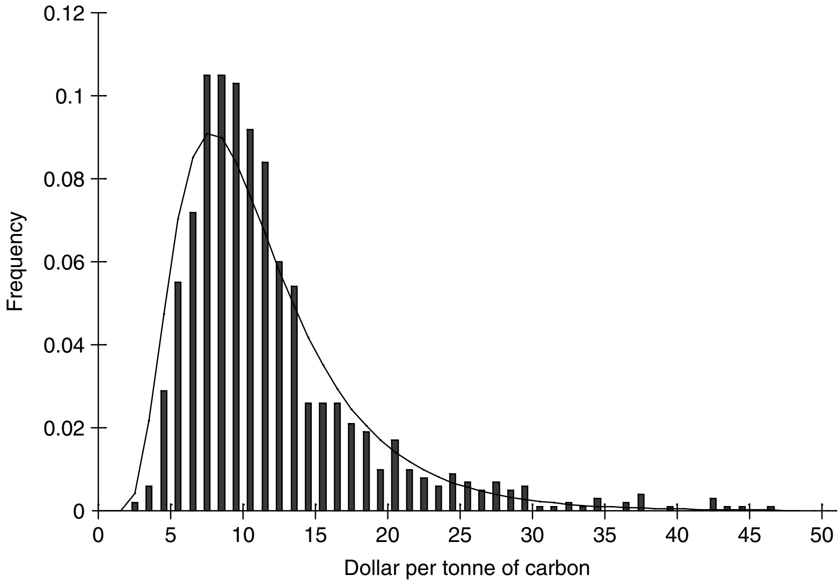
Notes:

[a] Pure rate of time preference, or utility discount rate. The more conventional consumption, or money discount rate equals the utility discount rate plus the growth rate of per capita income.

[b] Expected value, uncertainty about the discount rate included.

[c] Tol and Downing (2000) report estimates from Tol's *FUND* model (top line) and from Downing's *Open Framework* model (bottom line).

[d] Tol uses consumption discount rates of 1, 3 and 5 per cent; the assumed per capita income growth is roughly 2 per cent.



Source: Tol and Downing (2000).

Figure 14.2 Uncertainty about the marginal costs of carbon dioxide emissions for a 1 per cent PRTP and world average values density can be reasonably approximated with a lognormal distribution (the line in Figure 14.2).

14.5 CONCLUSIONS

The economic impact of climate change is a hard subject to study. Current methodologies are weak, and uncertainties remain large. Nonetheless, four conclusions can be drawn with some confidence.

First, vulnerabilities differ considerably between regions. Poorer countries would face proportionally higher negative impacts than richer countries.

Second, (sustainable) development may reduce overall vulnerability to climate change, as richer societies tend to be better able to adapt and their economies are less dependent on climate. But it is not known whether development will be fast enough to reduce poorer countries' vulnerability in time. Delays in reducing climate impacts could affect achievement of sustainable development targets.

Third, the impacts of moderate global warming (say, up to 2–3°C in 2100) are mixed. Poorer countries are likely to be net losers, richer countries (especially in mid- to northern latitudes) may gain from moderate warming. The global picture depends on how one aggregates. If aggregation is on a dollar basis, the world as a whole may win a bit. If aggregation is based on people, the world as a whole may lose. In addition, impacts to natural ecosystems could be negative even at these levels of warming.

Fourth, the impacts of more substantial global warming (more than 2–3°C or sooner than 2100) are probably negative, and increasingly so for higher or faster warming. This holds for the majority of countries. Note that, because of the slow rate of change in the energy sector, the atmosphere and the oceans, we are probably already committed to at least 2°C of warming.

It may be helpful to relate emerging relative confidence in climate change with our sense of progress in valuing climate change damages. Some climatic changes can be predicted with relatively high confidence – global and regional warming, sea level rise and rising CO_2 concentrations. These changes will affect, among other things, agroclimatic suitability, heat stress and demand for water. Less confidence is ascribed to changes in storm- and water-related effects: precipitation, precipitation intensity, wind speeds, sunshine and so on. However, the range of scenarios generally fall within defined limits, leading to modest confidence in expected impacts on crop production, water systems and other resources. Low confidence is likely to continue for some time in our ability to project changes in the risk of extreme events (prolonged drought, intense cyclones and so on) and large-scale changes such as collapse of major ice sheets. We have indicated above that confidence is higher in valuation of market impacts than non-market impacts, and equity and welfare effects are especially contentious.

What then can we conclude about the conjunction of confidence in climate change and the valuation of impacts (not ignoring uncertainties in GHG emissions and impacts science as well)? It does not take too much imagination to reach very large damages, but they require the incorporation of relatively uncertain climate changes and impacts, and the kinds of valuations of ecological and human systems that are not customary in present assessment models – that is, the lower and right-hand cells in the table. On the other hand, we have relatively high confidence that the market impacts of some trends in the climate system will have benefits in some regions and sectors (for example, northern agriculture) and costs in others (for example, coastal habitation in vulnerable deltas). The area in between these two poles, where climate futures are at best uncertain risks and valuation of non-market impacts is poorly understood, remains a fruitful research frontier.

Overall, there is a clear economic rationale for reducing greenhouse gas emissions. By how much, where and when cannot be answered without also considering the costs of emission reduction. One needs to compare the marginal damage costs of climate change to the marginal costs of emission reduction. The marginal damage costs of carbon dioxide emissions are uncertain, but the current literature suggests that estimates in excess of $50/tC require relatively unlikely scenarios of climate change, impact sensitivity and economic values.

NOTES

* This chapter draws heavily on the IPCC Third Assessment Report of Working Group II, chapter 19 (Smith et al., 2001). We are grateful to all the authors, reviewers and review editors of that chapter. The EU Directorate General Research (JOS3-CT95-0002, JOS3-CT97-0015), the US National Science Foundation through the Center for Integrated Study of the Human Dimensions of Global Change (SBR-9521914) and the Michael Otto Foundation provided welcome financial support. Tom Downing, David Pearce, Ari Rabl and Richard Richels provided helpful comments on an earlier version. The interpretations and any errors in this chapter are, however, our own and do not necessarily reflect the views of any of the institutions with which we are affiliated.

1. This is the best guess based on FUND2.0, using a 1 per cent pure rate of time preference, a 'years of life lost' valuation methodology for health risks and world average values for regional aggregation.

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15. Environmental expenditures

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15.1 INTRODUCING ENVIRONMENTAL EXPENDITURES[1]

As mentioned in Chapter 7, the definition of defensive expenditures was introduced by Leipert (1989) as 'an indicator of the total monetary burden which society bears annually for the regulation of environmental degradation and damages induced by the economic use of the environment in the past and present periods, including future oriented expenditures'. Broad categories of defensive expenditures are recognised as:

- environmental protection activities,
- environmental restoration/treatment of damages,
- damage avoidance from environmental deterioration.

This definition is broadly consistent with the SERIEE definition adopted by EUROSTAT in 1994, and incorporated recently into the SEEA revision. In this chapter monetary data are reported in Euro. Original figures in national currency were transformed into ECU using average exchange rates for 1994 or 1995, since calculations were made before 31 December 1998, when fixed exchange rates were not yet available. However, as an approximation, values are reported in Euro, as this approximation does not affect the substance of the chapter.

In the following data are reported that were available at the time of the research project: in the meantime data availability has improved significantly, thanks to the joint efforts of Eurostat and the OECD. At the end of the chapter (see Table 15.25) more recent data are displayed in order to show the advancement in this field and to provide a better picture of environmental expenditure in the EU-15 and not only in the case study countries.

15.2 ENVIRONMENTAL EXPENDITURE IN GERMANY

15.2.1 Introduction

In Section 15.2 data on defensive expenditures for environmental protection are presented according to the Federal Statistical Office (FSO) and consistency with the SERIEE classification is checked. In addition, defensive expenditures for specific purposes that are not considered by FSO are presented, such as the restoration of materials in historic and cultural buildings.

15.2.2 Defensive Expenditures for Environmental Protection according to the Federal Statistical Office

Data on environmental protection expenditure by the government and by private industry for air pollution control, waste water treatment, waste management and noise are collected by the Federal Statistical Office. These data also include aspects of damage restoration and treatment (for example, effluent treatment) and damage avoidance (for example, investment to achieve the critical loads set for air pollution in the Federal Immission Control Act). Data on both capital and current expenditures are available. Concerning private industry data cover end-of-pipe and clean technologies as well as investment in clean products. Public expenditure includes end-of-pipe technologies, nature conservation measures and research and development activities. Household expenditure includes estimates on catalytic converters and waste water charges. Expenditure on environmental monitoring and consultancy is included in the total expenditure figures for the key environmental domains, such as air pollution control, waste water treatment and waste management (ECOTEC/BIPE/IFO, 1997). Table 15.1 gives an overview of data covered by the Federal Statistical Office (except for household data, which are estimated).

In total, environmental expenditure calculated according to the above-mentioned categories amounts to about 32 billion Euro in 1994 or 2 per cent of German GDP (ECOTEC/BIPE/IFO, 1997; see also Table 15.2). The largest capital expenditures are undertaken by government, with most spent on waste water treatment. Industry undertakes more than 50 per cent of its environmental investment in air pollution control activites. This is mostly driven by strict environmental regulation. Household figures are hardly available in Germany; this explains the low amount of expenditure. A somewhat indirect form of defensive expenditure by households is membership fees paid to environmental associations. However, this is not

Table 15.1 Data coverage for Germany

Expenditure item	Public			Private			Households		
	Cap.	Op.	Total	Cap.	Op.	Total	Cap.	Op.	Total
End-of-pipe technologies[a]	✓	✓	✓	✓	✓	✓			
Clean technologies				✓					
Energy conservation measures									
Renewable energy technologies									
Nature conservation measures[b]	✓	✓	✓						
Environmental R&D	✓	✓	✓						
Savings from environmental activities									
Production/use of cleaner products/ materials[c]				✓			✓		
Environmental taxation[c]								✓	

Notes:
[a] Private operating expenditure only for West Germany.
[b] Included in 'other' category.
[c] Household expenditures are estimates.
Cap. = Capital expenditure. Op. = Operational expenditure.

Source: ECOTEC/BIPE/IFO (1997), German Country Profile, table 1.

reported in Table 15.2. The literature in this field is sparse; one study reports a volume of about 70 million DM, corresponding to approximately 35 million Euro (Hampicke, 1991).

During recent years a shift from capital to current expenditure has taken place. Fifty-six per cent of total expenditure in 1993 was operating expenditure. Within the various areas, however, there is variation. The waste management sector shows the highest share of operating expenditures (76 per cent), reflecting high transport and collection costs. Recycling efforts as part of waste management are becoming more and more important; they are connected to a shift in expenditure from the public to the private sector because private systems such as the DSD (Duales System Deutschland Gmbh) exist for the collection, sorting and recycling of packaging waste.

Table 15.2 Environmental expenditure in Germany (1994, million Euro)

Environmental domain	Public sector			Private Sector			Households			Total			% Tot
	Capital	Operat.	Total	Capital	Operat.	Total	Capital	Operat.	Total	Capital	Operat.	Total	
Air pollution control	263	470	733	2404	5349	7753	1354	0	1354	4022	5819	9840	31
Waste water treatment	6589	2088	8686	1461	4239	5701	0	87	87	8059	6414	14473	45
Waste management	1036	3582	4618	612	1545	2157	28	107	136	1676	5234	6911	22
Remediation	nr	nr	nr	nr	nr	nr	nr	nr	nr	nr	nr	nr	nr
Noise and vibration	nr	nr	nr	155	109	264	nr	nr	nr	155	109	264	1
R&D (not included in other domains)	7	209	215	nr	nr	nr	nr	nr	nr	7	209	215	1
Other	44	126	170	nr	nr	nr	nr	nr	nr	44	126	170	1
Total expenditure	7947	6475	14 422	4633	11 242	15 875	1383	194	1576	13 963	17 911	31 873	100

Notes:
Data/estimates refer to 1993. Currency year is 1994 using an inflator of 1.04 (EUROSTAT Yearbook 1995). 1994 ECU/DM exchange rate was 0.520833. Data confidence: percentage of data non-estimated = 99 per cent. Capital = Capital expenditure. Operat. = Operational Expenditure. Nr = not reported.

Source: ECOTEC/BIPE/IFO, (1997), German Country Profile, table CP1b, which is largely based on data collected by the Federal Statistical Office (1995).

The true economic cost of environmental protection activities is reflected in the depreciation of the respective capital stock. For West Germany this amounts to approximately 10 billion Euro in 1993 or 1.4 per cent of GDP. No data are available for East Germany (ECOTEC/BIPE/IFO, 1997).

Only some of the categories of the SERIEE classification (characteristic activities, connected products, adapted products, specific transfers) are covered by official statistics.

Table 15.1 shows that the approach of the German Federal Statistical Office does not cover all parts of the SERIEE categories for environmental expenditures. For instance, data on soil and groundwater pollution is not collected at the federal level in Germany (see also Section 2.4 on the restoration costs of contaminated sites). Data on the protection of biodiversity by households is missing altogether, although some work has been done within the framework of specific studies (see Chapter 11). Concerning households, there is also a lack of knowledge on expenditure for connected products. At present, only data on catalytic converters are available. Valid data on adapted products are also sparse despite the fact that, for example, some Länder (hereinafter states) have reinforced their green requirements in procurement. Finally, with respect to specific transfers, the official statistics cover the waste water charge which is earmarked for environmental purposes.

15.2.3 Defensive Expenditures for Materials in Historic and Cultural Buildings

Background

It is difficult to estimate the defensive expenditure involved in the restoration of historical buildings. In fact, it is extremely difficult to detect the part of the damage to be attributed to air pollution, establishing reliable exposure–response functions in this area.

The development of the first studies on the effect of pollutants on stone monuments reflects the fact that only in recent times has a distinction been made between particulate and gaseous pollutants (Briblecombe, 1989). More recently, studies on pollutant-induced deterioration have become more and more frequent. They aim to investigate in greater detail the mechanisms of stone/pollutant action and factors influencing the possible reactions. The availability of more sophisticated analytical techniques together with the traditional petrographic techniques has contributed to a better comprehension of the role played by the different pollutants – gaseous and particulate (Laurenzi Tabasso, 1992).

Deterioration is faster in urban environments than in unpolluted sites with similar climatic conditions. For example the estimated rate of surface loss for marble in rural sites does not exceed 0.5 mm every 100 years, while in urban environments the figures are more than ten times higher.

Lipfert (1989) determined the interaction between $CaCO_3$ (calcium carbonate), acid pollutant and rain on the basis of statistical data, suggesting the following function:

$$Surface - loss(\mu m/m_{\text{rain}}) = 18.8 + 0.016ph + 0.18\,V_d SO_2/R$$

where: Vd is the deposition rate of SO_2 (cm/s),
SO_2 is the concentration of SO_2 ($\mu m/m^3$) and
R is the amount of rain (m).

It should be noted that this function does not consider intermittent, catastrophic events, such as the detachment of surface crusts.

The effects of atmospheric pollution, as well as of acid rain, double the frequency of maintenance costs. Some authors (Baldi, 1989) have estimated that the frequency of maintenance for building facades was around 25 years before the Second World War. In the present situation facing many polluted town centres (under European climatic conditions) such maintenance work has a frequency of 7–8 years.

This is a field to explore further, in order to calculate the proportion of expenditure on monuments to be attributed to air pollution. A pragmatic solution could be to calculate for each country the percentage of expenditure which is to be attributed to air pollution on the basis of statistical evidence. This would be helpful for the purpose of environmental accounting.

According to the Law on the Protection of Historic Sites there is a wide definition of historic sites in Germany, including not only classical objects like churches, castles, townhalls and so on, but also areas of urban planning and housing from the nineteenth century, industrial sites of historical relevance, the large area of residential buildings and historic streets and places as well as historic city areas (Hummel and Waldkircher, 1992).

The number of historic buildings in West Germany is estimated to be about 462 000 of which the majority (about 75 per cent) are in private hands according to a study done for North-Rhine Westfalia in the mid-1980s (Deutsches Nationalkomitee für Denkmalschutz, 1983; Minister für Landes- und Stadtentwicklung des Landes Nordrhein-Westfalen, 1984).[2] The rest are owned by churches and the government in equal measure.

The average number of historic sites per municipality in West Germany is 264 in 1985. In large cities with more than 500 000 inhabitants the average number is 3385; in cities with more than 100 000 to 500 000 inhabitants on average 886 sites are reported to be historical monuments. This figure declines to 358 in cities of middle size and to 98 in smaller cities (Echter, 1987).

In East Germany, before reunification, there were lists of historic sites which were substantially extended after 1989. For political reasons many monuments and sites were not well protected in socialist times. Today there are about 68 000 historic buildings (BMU, 1992). Originally 180 East German towns were considered to be historic and their entire conservation was planned. This number is even higher today, but no exact figure is available. In former East Germany defensive expenditures were mainly devoted to the restoration of monuments of international importance.

Methodology

No Contingent Valuation Method (CVM) analysis has recently been undertaken in Germany in the area of materials. According to the International Council on Monuments and Sites (Munich) it is extremely difficult to establish reliable exposure–response functions in this area. Only the decay of metal monuments situated outside is relatively easy to describe. However, the decay of stone and nowadays of concrete is very hard to trace back to air pollution. Other possible sources of damage to cultural buildings, such as military aircraft noise, are even harder to determine. In one recent incident in Germany, a fresco painted 300 years ago by Georg Asam fell from the ceiling of a castle near Regensburg (*Süddeutsche Zeitung*, 1997). Allegedly military aircraft exercises have caused the damage, which it is estimated will cost 300 000 to 500 000 DM to repair (corresponding to Euro 153 374 to 255 623). However, no clear scientific proofs are available.

Due to these methodological problems the current study will only attempt to quantify defensive expenditures for the restoration of historic sites in general. No effort is made to isolate the purely environmental share of such expenditures.

Expenditures on the restoration of historic sites

Due to the immensely wide definition of historic sites it is very difficult to get a clear picture of the economic importance of historic sites. Data on defensive expenditures can only be obtained for public expenditures at federal and state level. The immediate net expenditures are 337 million Euro in 1993 (personnel costs: 25 million Euro; construction measures: 43 million Euro). More than 90 per cent of these expenditures are paid by the states. Transfers to other areas, which also include grants to private persons, were 141 million Euro in 1993 (Federal Statistical Office, 1996a).

Municipal expenditures are not available in German financial statistics. Only a collective term for ‘Other expenditures in the area of art and culture’ is available. In 1993 total expenditures under this category were 598 million Euro. However, this category also covers expenditures on

Table 15.3 *Total net expenditures for protection of historic sites in Germany in 1980, 1990, 1992 and 1993 (federal government and states)*

Total Net Expenditures for Protection of Historic Sites			
Year	Total (million Euro)	Federal government, ERP, LAF (million Euro)	States (million Euro)
1980	152	8	144
1990	217	6	211
1992	312	35	277
1993	337	31	306

Source: Federal Statistical Office (1996a).

Table 15.4 *Type of expenditures for protection of historic sites in Germany (Federal government and states) in 1980, 1988 and 1993*

Type of expenditure (million Euro)	1980	1988	1993*
Net expenditure	152	224	337
Immediate expenditure, of which	128	190	240
● personnel	8	19	25
● construction	44	46	43
● transfers	60	103	141

Note: * Includes East Germany.

Sources: Federal Statistical Office, several years.

landscape preservation (among other things). The data on expenditures for historic sites are summarised in Tables 15.3 to 15.5.

It is even more difficult to estimate the expenditures made by private persons without government grants. One approximation could be found in the amount of tax deductions for the conservation of historic monuments found in income tax statistics (specifically §§ 82i and 82k of the German income tax ordinance). Because the income statistical data reported here date back to 1989, values in Table 15.6 refer only to West Germany.

In 1991, §§ 82i and 82k of the German income tax ordinance were changed into § 7i of the German law on income tax. According to this new paragraph, the private cost for the conservation of (legally defined) historic

Table 15.5 Expenditures for protection of historic sites in federal states in 1993

Federal state	Net expenditures (million Euro)	% of total
East Germany	87	28.0
Mecklenburg-Vorpommern	10	3.2
Saxony	29	9.3
Saxony-Anhalt	14	4.5
Thuringia	3	0.8
Brandenburg	31	10.2
West Germany	219	72.0
Bavaria	58	19.2
Baden-Württemberg	51	16.7
Berlin	19	6.2
Hessen	18	6.0
Rhineland-Palatinate	10	3.2
Saarland	2	0.7
Lower Saxony	15	4.9
Schleswig-Holstein	5	1.7
North Rhine-Westfalia	37	12.1
Hamburg	3	1.0
Bremen	1	0.3
Germany	306	100

Source: Federal Statistical Office (1996a).

monuments can still be deducted from income tax. According to the Ministry of Finance, reduced income tax revenues both in East and West Germany from 1991 to 1994 correspond to Euro 26 million, of which Euro 11 million are borne by the federal government.

The need for private expenditures seems to be much higher than the actual expenditures can convey. In 1996 the Housing Ministry estimated a potential need for defensive expenditures in the area of historic and cultural buildings for both East and West Germany of about Euro 5–6 billion (Petzet, 1997).

Next to private and public expenditures, there are four important foundations in Germany which make a major contribution to the conservation of historic monuments:

1. Deutsche Bundesstiftung Umwelt;
2. Deutsche Stiftung Denkmalschutz;

Table 15.6 Tax deductions for the conservation of historic sites

Year	Total tax deductions (million Euro)
1980	16 563
1983	38 291
1986	60 829
1989	55 288

Source: Federal Statistical Office (1996b).

Table 15.7 Defensive expenditures provided by the Deutsche Bundesstiftung Umwelt

Year	Total expenditures in million Euro
1991	27
1992	10
1993	34
1994	15
1995	13
1996	7

Source: Deutsche Bundesstiftung Umwelt (1997).

3. Messerschmidt Stiftung;
4. Wüstenrot-Stiftung.

The *Deutsche Bundesstiftung Umwelt* supports pilot projects which help to restore historic monuments damaged by pollution.[3] The expenditures shown in Table 15.7 include not only the foundation's payment appropriations for the period between 1991 and 1996, but also the costs borne by the applicants, that is, the figures represent total costs.

In 1996 the *Deutsche Stiftung Denkmalschutz* supported 236 projects in Germany with a payment appropriation of Euro 17 million. Since its foundation in 1985 until 1996, it has funded about 845 monuments, comprising:

- 348 churches,
- 223 town houses,
- 34 monasteries,
- 113 forts and castles,

- 34 technical monuments,
- 32 historic city halls and theatres,
- 8 gardens and parks.

The *Messerschmidt Stiftung* was not able to provide any figures.

After the German reunification the *Wüstenrot-Stiftung* developed a programme for the renovation and revitalisation of historic sites in East Germany. Since 1993 the foundation has spent about Euro 10 million on the restoration of about 15 splendid monuments of international importance (Wüstenrot-Stiftung, 1997). The foundation does not provide grants, but acts in an operative manner. The foundation's criteria for taking action are the following:

- The monument to be restored must be in public hands.
- Funding will only be provided for splendid monuments of international importance.
- The foundation undertakes operative business, that is, it does not only provide the necessary resources, but also takes care of expertise on renovation strategies, contracts with architects and supervises the construction work.
- Finally all monuments must be of general cultural interest and accessible to the public.

There is another important foundation, the *Volkswagenstiftung*, which however does not fund the actual costs of renovation or conservation of historic monuments, but finances scientific expertise prior to the conservation process. Currently the foundation finances an inventorisation project called MIDAS (developed at the University of Münster), of which the purpose is to build an inventory of historic monuments in the new states (Volkswagenstiftung, 1997).

With respect to the available data on defensive expenditures (for example, no aggregate data could be obtained from the Church, which is also active in the area of conservation of historic sites) the following per capita figures and shares in GNP can be calculated (Table 15.8).[4]

15.3 ENVIRONMENTAL EXPENDITURE IN ITALY

15.3.1 Environmental Expenditure in the Public Sector

Several studies have been carried out in Italy regarding environmental expenditures, particularly concerning public expenditure. The interest in

Table 15.8 Defensive expenditures as per capita figures and shares in GNP in Germany in 1993

Type of expenditure	Total expenditures in million Euro	Total expenditures in % of GNP	Total expenditures per capita in Euro
Federal and state level	337	0.03	4.2
Private households	26	0.0021	0.3
Foundations	56	0.0046	0.7

Source: Calculations of Ifo Institute, 1998.

this topic is quite recent, the first few studies being produced in the mid-1980s. While theoretical research in this field is going further, practical applications still remain incomplete and less rigorous than theoretical advancements. Major difficulties derive from lack of data collection and monitoring. These stem from:

- the high number of public actors involved in environmental monitoring and data collection at different levels;
- inconsistencies in balance sheet criteria within public administrations;
- the high number of financial transfers among different public institutions and the difficulty of recording them in a complete and correct way, avoiding double counting.

Official statistics on environmental expenditure can be found both in the Report on the State of the Environment published by the Ministry of the Environment and in the statistical periodicals published by ISTAT (usually every two years). The first set of data mainly refers to allocation of resources by law and therefore data are not presented according to a 'well-established' classification of environmental expenditure. Data published by ISTAT are more consistent with the SERIEE system, while those published by the Ministry of the Environment tend to refer to the Italian laws and programmes classification. Therefore, below we will almost exclusively refer to the ISTAT source of data. From the late 1980s the Ministry for the Environment gave a mandate to the ISPE research institute (Istituto per gli Studi di Programmazione Economica) to estimate public environmental expenditure at national level. Two main studies were produced: the first in 1991 and the second in 1996. Results are published in ISTAT publications, such as *Environmental Statistics*.

Table 15.9 State environmental expenditure: breakdown by resources (million Euro)

Year	Land	Water	Air	Others	Total	Rate of increase
1986	468.58	5.01	58.77	2.12	534.48	
1987	437.80	17.40	60.12	4.13	519.45	−0.028
1988	424.58	49.94	139.60	17.87	631.99	0.217
1989	661.53	72.25	69.46	28.77	832.01	0.316
1990	384.50	165.06	358.21	32.38	940.21	0.130
1991	1170.70	70.91	258.28	23.29	1523.19	0.620
1992	543.21	250.84	137.84	139.39	1071.29	−0.297
1993	712.40	33.88	112.59	61.97	920.84	−0.140
1994	780.68	99.93	147.96	164.96	1193.53	0.296

Source: Adapted from Cesaretti (1996).

Official statistical data on state environmental expenditure

As already mentioned, the ISPE study is the most complete and recent work on state environmental expenditure for our purposes. Moreover, it constitutes an official reference in the Italian context, as it appears in official statistical publications (ISTAT, 1996 and EUROSTAT, 1996c). This study contains data from 1986 up to 1994. Some difficulties still remain in classifying environmental expenditure according to the EPEA classification, as the Italian state and regions adopt the expenditure classification introduced by the Ministry for the Environment according to law 305/89. This attributes expenditures to environmental media (air, water, soil), thereby reflecting a prevailing *command and control* approach. The ISPE study represents the most useful attempt to classify expenditure in a more disaggregated way, although it is not completely consistent with the SERIEE system.

The ISPE study refers to different types of classifications: on the one hand to a classification by resource (land, air and water), and on the other, to an economic classification into current and capital expenditure, as well as by economic sector.

Table 15.9 shows state expenditure according to classification by resource and the overall trend of the expenditure. This table shows that state environmental expenditure has increased substantially from the late 1980s/early 1990s for all media and soil in particular. It increased constantly from 1988 until 1991, but in 1992 decreased by 29 per cent and in 1993 by 14 per cent. However, an increase (of 29 per cent) has been recorded in 1994.

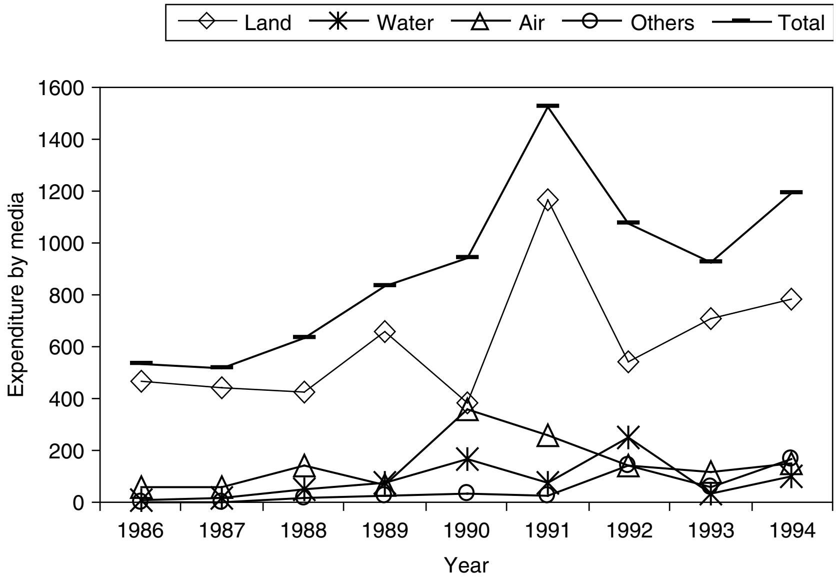

Figure 15.1 State environmental expenditure: breakdown by media (million Euro)

Figure. 15.1 shows the trend of state expenditure by media.

In Table 15.10 figures are broken down according to a more detailed classification, albeit different from the SERIEE classification, as has been pointed out.

In Italy state environmental expenditure is borne by several Ministries. Since 1986, when the Minister of the Environment was appointed, the expenditure borne by this Ministry has increased sharply. Environmental expenditure borne by the Ministry of the Environment represented approximately 0.02 per cent of the total environmental expenditure in the first year of his appointment, while in 1992 it represented 43.7 per cent of total state environmental expenditure. The three-year environmental plan published in 1993 stressed the importance of integrating the environment into other policies, as suggested by the fifth Environmental Action Programme of the EU in force at that time. This is probably the reason why environmental expenditure borne by the Ministry of Environment recorded a decrease afterwards. In fact, other Ministries, such as Treasury, Budget and Economic Planning, Public Works, Agriculture, Industry, Transport, Health, Foreign Affairs, Scientific Research, and even the Prime Minister's Office contribute significantly to the state environmental expenditure.

Table 15.10 State environmental expenditure: breakdown by resources and activity (million Euro)

Category	1986	1987	1988	1989	1990	1991	1992	1993	1994
Forest and mountain systems	169.19	185.92	195.12	376.34	73.80	449.78	223.94	339.93	293.35
Hydrographic systems	276.87	182.46	171.83	221.71	188.29	593.51	149.72	185.77	295.16
Coasts and lagoon systems	16.22	31.61	30.73	21.38	49.27	52.68	64.25	76.18	56.29
Wetlands subsidence and others	3.25	4.96	3.72	10.48	27.27	20.30	30.73	42.30	22.62
Land reserves and parks	0.41	25.41	14.82	15.80	17.04	17.15	30.37	29.90	35.48
Waste disposal		–	–	–	9.50	7.33	20.92	4.34	2.58
Industrial site relocation and restoration							–	3.62	5.16
Restructuring of agriculture		4.13	3.10	9.30	7.95	6.87	8.01	1.29	7.28
Instrumental measures	2.89	3.31	5.27	6.51	11.36	23.09	15.29	29.08	62.75
Land	468.84	437.80	424.58	661.53	384.49	1170.70	543.21	712.40	780.68
Inland waters	0.98	–	1.19	10.02	50.41	28.77	68.02	13.12	66.21
Marine waters	3.82	17.20	47.82	60.68	112.59	38.89	180.50	17.51	26.39
Aquatic reserves and parks	–	–	0.83	1.39	2.01	3.10	0.72	2.17	2.53
Instrumental measures	0.21	0.21	0.10	0.15	0.05	0.15	1.60	1.08	4.80
Water	5.01	17.40	49.94	72.25	165.06	70.91	250.84	33.88	99.93

Table 15.10 (continued)

Category	1986	1987	1988	1989	1990	1991	1992	1993	1994
Energy saving and renewable resources	58.77	60.12	139.60	69.21	357.80	256.06	56.14	105.62	140.37
Reduction of emission and noise				–	–	–	78.55	2.27	5.58
Instrumental measures	–	–	–	0.26	0.41	2.22	3.15	4.70	2.01
Air	58.77	60.12	139.60	69.46	358.21	258.28	137.84	112.59	147.96
Other (not classified)	2.12	4.13	17.87	28.77	32.38	23.29	139.39	61.97	164.96
Total	534.74	519.45	631.99	832.01	940.15	1523.19	1071.29	920.84	1193.53

Source: Cesaretti (1996).

It is important to note that actual expenditure represents a small percentage in comparison with the total expenditure capacity (*massa spendibile*), comprising allocated resources for the year examined as well as resources left over from previous years. In fact, a large part of these resources are left over because procedures for allocating resources are not completed within the year and because often recipients do not realise projects once resources have been allocated. The average percentage of actual expenditure compared with the total expenditure capacity for the period 1986–94 was 22.4 per cent.

These data do not distinguish between current expenditure and investment, but the updated Report on the State of the Environment in Italy (Ministero dell'Ambiente, 1997) pointed out that investments prevail over current expenditure in terms of amount.

In the last few years (1991–95) the average investment rate represented 84.8 per cent of total state expenditure, whereas in a study carried out by Padua University (Bruzzo et al., 1994) for 1987–89 the investment rate represented 63.9 per cent. The work by Bruzzo et al. (1994) also pointed out that at that time most investments were traditional, such as water treatment, waste disposal, forestation policy and so on. The significant increase in environmental investments is probably due to the positive effects on planning caused by the first and the second three-year programmes.

The most recent study highlights a high investment rate, but figures are not calculated on the basis of the SERIEE approach. In fact, current expenditure does not include personnel and administrative costs, which are taken into account in the SERIEE classification. This could lead to an underestimation of current costs and therefore to an overestimation of the investment rate. In fact, the study by the University of Padua, which considers part of the personnel costs, shows a lower investment rate. On the other hand, other expenses are taken into account in the classification of the Italian Ministry for the Environment which are not included in the SERIEE approach. This explains why the figures presented above do not match those of the ISPE institute, the latter being the most consistent with the SERIEE classification.

Estimates on total public environmental expenditure

In the framework of public environmental expenditure we should consider state expenditure, as well as regional and local expenditure. The ISPE's work on state expenditure only takes into account direct state expenditure and transfers to regions, local authorities, households, industry and other institutions.

There are two drawbacks with this approach: on the one hand those transfers do not necessarily correspond to effective expenses; on the other hand this approach does not consider total public expenditure, inclusive of the total direct expenditure borne by regions as well as by local authorities.

The study by the University of Padua constitutes the first Italian attempt to evaluate total public environmental expenditure. This methodology avoids double counting, calculating direct expenses for each institution and then proceeds with the consolidation of these results. Unfortunately, this study refers to the years 1987–89 and is therefore somewhat dated.

The classification of environmental expenditure used in the University of Padua study differs from the one used in the ISPE study (Cesaretti, 1996) and adopted by the Italian Statistical Office. Therefore, it is difficult to compare these data with those published by ISTAT, the national statistical office. However, the main contribution of the University of Padua study consists in providing percentages of the expenditure borne by regions and municipalities, which allows us to estimate the total environmental expenditure of the whole public sector for recent years.

Table 15.11 gives a rough idea of the total environmental expenditure of the public sector. From the study of Bruzzo et al. (1994) we calculated that for the years 1987–89 the direct environmental expenditure of regional and local government is on average 11.7 times (rate of adjustment) state direct expenditure. Therefore, we can adjust data provided in Table 15.9 taking into account the expenditure of regions and municipalities. As a result, for the year 1994, which constitutes our representative year, total environmental expenditure for the public sector is more than 6 billion Euro. This leads to the conclusion that official data underestimate considerably the efforts made by the public sector. These are estimated to be on average 4.3 times

Table 15.11 Estimated environmental expenditure of the public sector (million Euro)

Year	State direct expenditure	Estimated regional and local gov. exp.	Total public expenditure
1986	227.29	2659.32	2886.62
1987	256.73	3003.75	3260.48
1988	304.66	3564.50	3869.16
1989	279.97	3275.66	3555.64
1990	368.91	4316.19	4685.10
1991	762.03	8915.78	9677.81
1992	340.29	3981.43	4321.73
1993	392.87	4596.56	4989.43
1994	491.56	5751.30	6242.86

Source: Adapted from Cesaretti (1996) and Bruzzo et al. (1994).

higher than official data. In fact, available data only refer to state direct expenditure and to that part of the regional and local government expenditure consisting in transfers coming from the national government.

However, it should be noted that the rate of adjustment used for these estimates is based on a less detailed classification.[5] The main difference seems to be that the ISPE classification (Cesaretti, 1996) does not take into account important expenditure such as sewage networks and the collection of waste which are included in the University of Padua study (Bruzzo et al., 1994). This may be due to the fact that this kind of expenditure originates at local level and is probably detectable at that level only. However, this expenditure is included in the SERIEE classification. Therefore, the adjustment rate seems to mitigate the underestimation caused by official statistics, which are based on the ISPE approach.

As a conclusion, we can state that estimated environmental expenditure for the whole public sector consisted of more than 6 billion Euro for the year 1994, which is rather low if compared to Germany and the United Kingdom.

15.3.2 Environmental Expenditure by the Private Sector

Data on environmental expenditure for the *industrial sector* is collected regularly through a periodical census. However, information collected through the census is not adequate for our purpose since data on environmental expenditure is presented in a very aggregated form. In fact, the census includes one only question pertaining to environmental expenditure. Moreover, it is addressed to companies with more than 20 employees, that is, small enterprises are excluded.

In Table 15.12 official statistics on environmental expenditure are presented for the period 1989–92. These figures are not able to capture defensive expenditure as defined by the CEPA classification. However, a more detailed survey was carried out by ISTAT in 1986, recording not only the

Table 15.12 Current environmental expenditure by industry (million Euro)

	1989	1990	1991	1992	Percentage (1989–92)
Current environmental expenditure	391	524	545	629	60.2
Env. exp./no. of employees	96	125	133	152	59.2
Coverage of the census	60.2	68.4	62	53.9	

Source: Battellini and Taccini (1996).

Table 15.13 Environmental expenditure in industry: 1986 survey (million Euro)

	Investment expenditure	Current expenditure	Investment/ current exp.	Total
Waste water treatment	55	136	0.40	191
Waste disposal	17	98	0.17	114
Air pollution emissions abatment	70	73	0.96	143
Total	141	306	0.46	447

Source: Battellini and Taccini (1996).

current expenditure but also investment expenditure, distinguishing between waste water treatment, waste disposal and air pollution emissions abatement. This survey divided environmental expenditure by sector. For our purposes the interest of this survey consists in considering both investment expenditure and current expenditure, as well as in providing figures in a more disaggregated way. In Table 15.13 the figures are broken down into three different types of expenditure.

All existing estimates on environmental expenditure for industry are based on this survey. For example, Carlucci (1990) estimated the environmental expenditure for industry for 1985 and 1987 assuming that the proportion of the expenditure for investment is fixed (0.46 per cent) and using the existing data on current expenditure. The same methodology was applied by Cullino (1992, 1996) for the years 1986–88. However, some slight differences in calculations need to be pointed out. These are due to differences in rounding up. In addition, Cullino as well as Falcitelli (1994) estimated expenditure on research and development in the environmental field using existing statistical sources recording expenditure on research using data available at Confindustria, the national industrial association. These estimates allowed us to construct a time series, which takes into account both official statistics and additional studies (see Table 15.14).

It would be useful for companies to adopt the CEPA classification in their accounting systems. For this purpose FEEM has developed a software programme of environmental accounting for companies which has as a starting point the CEPA classification (Bartolomeo et al., 1995). At the moment only a few large companies use this classification for their environmental accounts.

Table 15.14 Environmental expenditure by industry: official statistics and surveys (million Euro)

Year	Current expenditure		Investment expenditure		Total expenditure	
	Official statistics	Estimates	Official statistics	Estimates	Official statistics	Estimates
1985		269.59 (3)		123.95 (3)		393.54 (3)
1986	305.23 (1)		140.99 (1)		446.22 (1)	
1987		365.91 (4)		168.62 (4)		534.53 (4)
1988		477.21 (5)		229.05 (5)		706.25 (5)
1989	333.63 (2)	629.56 (6)		293.35 (6)		922.91 (6)
1990	448.28 (2)	705.48 (6)		330.02 (6)		1035.50 (6)
1991	456.55 (2)					
1992	515.42 (2)					

Notes:
(1) Survey carried out by ISTAT in 1986.
(2) Official statistics from census, collected by ISTAT.
(3) Estimates from Carlucci (1990).
(4) Average estimate calculated on the basis of the studies by Carlucci (1990) and Cullino (1996).
(5) Average estimate calculated on the basis of the studies by Cullino (1996) and Falcitelli (1994).
(6) Estimates of Falcitelli (1994).

Source: Adapted from ISTAT (1996).

In the private sector we should include *households*. However, in Italy statistics on environmental expenditure by households are not available. The only study we can identify on household expenditure is the one undertaken by Cullino (1996), who tried to estimate the total expenditure of the public sector, industry and households. For completeness, all estimates are presented in Table 15.15. The expenditure of households registers the following items: mineral water, house maintenance and repair, clothes washing, clothing, health expenditure. These figures are not comprehensive of all defensive expenditure borne by households and are not recent, but it is worth mentioning them since they represent the only available figures.

There is an urgent need for more reliable and recent data on private defensive expenditure, particularly in official statistics. However, in the near future we understand that Italian institutions will give priority to the public sector. Industry data provided are an underestimation as they refer to companies with more than 20 employees. Data on households should only be considered a first attempt to estimate household expenditure.

Table 15.15 Total environmental expenditure by the public and private sectors (million Euro)

Public sector (millions of Euro)	1986	1987	1988
Environmental expenditure	2931	3603	4072
Defensive expenditure for health	386	437	502
Total defensive expenditure	3317	4040	4574
Defensive expenditure as % of GDP	0.71	0.8	0.82
Private sector (millions of Euro)	1986	1987	1988
Industry	497	606	669
Households	2603	3163	3584

Source: Cullino (1992).

15.3.3 Further Developments

At the national level ISTAT is working on different issues:

1. calculation of the total public defensive expenditure;
2. calculation of private defensive expenditure both for industry and for households.

At national level the problem of public expenditure is a major concern. In fact, ISTAT is working on the disaggregation of public environmental expenditure according to the CEPA classification for the purpose of the SERIEE system. But major differences are highlighted when comparing the European classification with the classification used in balance sheets by the Italian public administration. As already pointed out, classifications adopted by the public administration are different and less detailed than the CEPA classification. In addition different levels of administration do not use the same classifications, resulting in problems of comparability and aggregation.

ISTAT is planning new activities for estimating environmental expenditure by industry and households. In the near future data concerning industry will be collected through census, though these surveys will not apply the CEPA classification or include a breakdown by media.

Finally, the results of a pilot project addressed to households will be available soon, though it is not likely to use detailed classifications.

15.4 ENVIRONMENTAL EXPENDITURE IN THE NETHERLANDS

15.4.1 Methodology Used: Scope and Limitations

Statistics on environmental expenditures of the public sector, private firms and households are constructed by the Central Bureau of Statistics (CBS). They are periodically published in the CBS publication *Costs and Financing of Environmental Management* and in recent years in the annual *Environmental Balance* of the RIVM. These statistics refer to expenditures made with the intention of protecting, restoring or improving the state of the environment. A distinction is made between the 'costs of own environmental activities' of a sector and the 'net environmental burden' of a sector. The first consist of the interest and depreciation on environmental investments and of current expenditure on environmental activities. The net environmental burden is obtained by correcting current expenditure for transfers from one sector to another as shown in Table 15.16. Correcting for these transfers gives a more accurate picture of the costs actually borne by a sector.

The environmental cost statistics, as constructed by CBS, are limited in scope. To appreciate their significance, it is important to understand which costs are *not* included. The CBS definition of environmental costs excludes the following types of costs:

1. Costs of measures aimed at solving *internal* problems with the use of environmental resources, for example within firms, are excluded. An example is the purification costs of drinking water companies: these costs are not included in environmental cost statistics.
2. If expenditure on an environmental measure is recovered by the sale of by-products or by a more efficient use of the resource, no costs are attributed to the measure. Profitable environmental measures are therefore outside the scope of these statistics.

Table 15.16 Calculation of the net environmental burden of a sector

Costs of own environmental activities	
Paid levies, subsidies and payments for environmental services by third parties	+
Received levies, subsidies and payments for environmental services performed for third parties	−
Net environmental burden	

3. The damage cost of environmental pollution is outside the scope of these statistics.
4. The costs of goods and services that substitute for lost environmental services are outside the scope of these statistics.
5. If an activity is not carried out or terminated because of environmental reasons, this sacrifice is outside the scope of these statistics.

The scope of the environmental cost statistics is very narrow when, for example, compared with the definition of defensive expenditure as proposed by Leipert (1989). The definition of environmental cost statistics includes the costs of non-profitable measures that aim at protecting, restoring and improving the quality of the environment. Outside its scope are profitable measures and all costs (internal measures, damage costs, substitute goods and services, loss of output/activity) that are due to environmental pollution and degradation. Also outside the scope of the environmental cost statistics are the costs of nature and landscape protection. These costs are included in separate statistics: the costs and financing of the management of nature and landscape.

Even within the narrow scope of the environmental cost statistics, there are some areas where data are missing (CBS, 1996):

- the entire area of external safety;
- the relocation of firms for environmental reasons;
- environmental activities of households other than the use of low-sulphur domestic fuel oil;
- environmental activities in certain service sectors and small firms (less than 20 employees);
- environmental activities of port authorities.

CBS is making ongoing efforts to improve the quality and coverage of its environmental cost statistics.

15.4.2 Breakdown by Sector and by Environmental Theme

The costs of own activities and the net burden can be attributed to different sectors. The following sectors are distinguished in Table 15.17.

Agriculture Environmental activities in agriculture are primarily aimed at the prevention of the over-application of manure on land and at the reduction of emissions of ammonia. They also include measures to reduce nuisance due to noise and odour.

Table 15.17 Environmental costs and burdens of different sectors, 1985–97 (in billion Euro, constant 1998 prices)

	Environmental costs per sector				Financial burdens per sector			
Sector	1985	1990	1995	1997	1985	1990	1995	1997
Agriculture	0.0	0.1	0.5	0.5	0.0	0.1	0.5	0.5
Industry	0.8	1.3	1.7	2.0	0.9	1.4	1.8	2.2
Public utility	0.1	0.3	0.5	0.5	0.1	0.4	0.5	0.5
Construction	0.2	0.3	0.4	0.4	0.3	0.4	0.5	0.5
Trade and services	0.1	0.2	0.5	0.5	0.2	0.5	0.7	0.8
Traffic	0.1	0.2	0.3	0.3	0.2	0.3	0.3	0.4
Households	0.0	0.3	0.2	0.2	0.9	1.5	2.1	2.4
Central government	0.5	0.6	1.0	1.1	0.5	0.4	1.1	1.3
Other governments	1.7	2.3	3.3	3.6	0.5	0.8	0.9	0.8
Total	3.7	5.6	8.4	9.2	3.7	5.6	8.4	9.2
% GDP	1.6	2.1	2.8	2.9	1.6	2.1	2.8	2.9

Source: RIVM (1998).

Industrial firms, public utilities, construction industry and trade and services Industries and public utilities carry out environmental activities such as the purification of waste water and the scrubbing of waste gases, the prevention and sanitation of soil pollution. They also invest in environmental management systems, in the preparation of environmental permits and in research into clean production methods.

Traffic and transport Environmental activities in traffic and transport concern adjustments in vehicles and fuels that reduce air pollution and noise nuisance.

Households Households carry out few environmental activities that are included in the environmental cost calculation. Activities such as saving energy, cutting back on polluting activities or buying 'green' consumption goods are for one or more of the reasons explained above excluded from the calculations. The only environmental activity of households that is included is the use of low-sulphur domestic fuel oil. There are many activities, however, that are carried out for households by governments and their agencies and by private providers of environmental services. Households pay charges and levies for these services.

Central government The main tasks of central government (including the national institute of public health and environmental protection (RIVM), in the area of environmental management is the formulation of rules and targets for the other sectors and their enforcement. Government carries out policy-relevant research and contracts research from institutes and firms. Government supports and stimulates environmental activities in other sectors and collects several environmental levies and charges.

Other governments Other governments are provinces, water boards, municipalities and intermunicipal corporations. The twelve provinces are responsible for the operation of several environmental laws, including the law on waste management, the law on water pollution and the law on soil protection. With the exception of two provinces, they have delegated water quality management to water boards. To finance their environmental activities, provinces receive revenues from levies (especially the water pollution levy), payments for environmental services provided and they receive a contribution from central government.

The water boards (including specific water purification boards) have a delegated responsibility for surface water quality management (alongside their traditional responsibility for quantitative water management). They construct and operate water purification plants and monitor the discharge of effluents from other sectors. These activities are financed by the water pollution charge.

The municipalities are involved in the implementation and enforcement of environmental permits. They are responsible for urban waste collection and disposal and they also carry out an important part of the national noise reduction programme. Their environmental activities are financed by waste collection charges, sewerage charges and in some cases by water pollution charges. They also receive a contribution from central government.

Intermunicipal corporations have been established by government bodies to co-operatively carry out certain specific tasks, such as urban waste collection, disposal and/or incineration. Certain municipalities have delegated their permit management tasks to a regional intermunicipal corporation.

Table 15.17 reports the 'costs of own activities' and the 'net burdens' of nine sectors in the period 1985–97. It also shows total environmental costs as a percentage of gross domestic product.

Table 15.18 Environmental costs per environmental theme, 1985–97 (in millions of Euro, constant 1998 prices)

	1985	1990	1995	1997
Acidification	189	603	885	998
Climate change	66	90	224	325
Eutrophication	75	186	316	361
Dispersion	768	1178	1680	1887
Removal	1613	2193	3112	3380
Disruption	203	300	433	435
R&D	265	275	425	445
Implementation and enforcement	290	518	664	655
Soil sanitation	139	236	516	624
Others	31	26	110	135
Total	3638	5605	8364	9243

Source: RIVM (1998).

In the Netherlands environmental policy a number of environmental themes are distinguished. Below, the themes that are distinguished in the Third National Environmental Policy Plan are briefly described. Table 15.18 summarises the results for the different themes:

1. Acidification concerns the emission and deposition of acidifying pollutants such as SO_2, NO_x and NH_3.
2. Climate change concerns global warming caused by the emission of greenhouse gases, such as CO_2, CH_4 and N_2O.
3. Eutrophication concerns the eutrophication of surface waters caused by the emission of nutrients such as N and P to soil and surface waters.
4. Dispersion concerns the emission of hazardous substances to air, water and soil. This theme includes the emission of heavy metals, PAHs, PCBs, PM_{10}, ozone and pesticides.
5. Removal concerns the collection, processing and disposal of urban and industrial solid wastes.
6. Disturbance concerns all kinds of disturbance at the local level, including noise and odour, external safety and urban air pollution.
7. Research and development (R&D).
8. Implementation and enforcement.
9. Soil sanitation.

10. Others include the protection of the ozone layer and the protection of natural areas against dehydration due to groundwater extraction and groundwater table management.

15.4.3 Nature and Landscape Management

Table 15.19 presents data on the costs of nature and landscape management for different sectors. These data are extracted from the statistics 'costs and financing of the management of nature and landscape' that are published by the Dutch Statistical Office (CBS).

15.4.4 Conclusions

Environmental costs, as defined by CBS, are limited in scope. They are part of the broader concept of defensive expenditures, but they do not cover all its aspects. Nevertheless, environmental costs, as defined by CBS, have increased to Euro 9.2 billion in 1997 or to 2.9 per cent of GDP. In real terms, environmental costs increased by a factor of 2.5 in the period 1985–97. Environmental expenditures account for 1.4 per cent of government spending. Transfer payments may cause a large difference between the costs of a sector's environmental activities and its ultimate financial burden. The largest difference can be found in the households sector which carries out few environmental activities but pays many levies and charges. Households and industry together carry about half of the environmental burden. In terms of environmental themes, dispersion (hazardous substances, sewerage) and removal (urban solid waste), taken together, account

Table 15.19 Costs of nature and landscape management (million Euro, current prices)

	1991	1993	1995
Central government	176	185	219
Provinces	43	55	54
Water boards	3	3	2
Municipalities	14	21	25
Firms	32	36	51
Private forest owners	22	24	23
Nature conservation societies	63	67	77
Unallocated	3	3	4
Total	356	394	455

Source: CBS (1996, 1998).

for more than half of the costs. The costs of nature and landscape management were 0.5 billion Euro in 1995. About 65 per cent of these costs are financed by government, the remaining 35 per cent is paid by private parties.

15.5 UK RESULTS

15.5.1 Introduction

UK experience in the measurement of defensive expenditures is based heavily on survey data. There are, at present, no official statistics on this subject and it appears that the UK government feels that it is prudent to establish sound methodologies before embarking upon official data collection exercises. The government research effort to date has concentrated mostly on the industrial sector and the results from this research are presented below. Research on the household and government sectors has been undertaken by a group at Keele University, funded by the World Wildlife Fund (WWF) (Simon, 1997). Their results are also reviewed below.

15.5.2 Industry Expenditures

The UK government has commissioned a series of surveys of environmental spending by companies in the extraction, manufacturing, energy and water supply industries (DoE, 1996; DETR, 1998, 2001). Definitions of environmental expenditures adopted in these surveys were those given by EUROSTAT. This section will review the results of the 1994 and 1997 studies (DoE, 1996; DETR, 1998).[6]

Table 15.20 shows environmental expenditure by media. Control of liquid discharges makes up 31 per cent of total environmental expenditures, whilst air and solid wastes account for 25 per cent and 28 per cent respectively. The largest proportion of expenditures is for operating expenses.

Table 15.21 provides an overview of the environmental expenditure made by industry in the United Kingdom. This shows that the largest expenditure in 1997 was in the chemical industry, amounting to 24.3 per cent of the total industrial expenditure on environmental protection in the UK. DETR (1998) also found that larger firms with over 250 employees accounted for over two-thirds of total environmental expenditure, though they only made up 2 per cent of the sample.

These results can be compared with those for 1994, as shown in Table 15.22. As can be seen from the table, the largest rise in environmental expenditure was in the coke, refined petroleum and nuclear fuels sector,

Table 15.20 Environmental expenditure by industry by media in the UK, 1997 (Euro 1995)

Environmental media	Capital expenditure		Operating expenditure		Total expenditure	
	million Euro	%	million Euro	%	million Euro	%
Liquid discharges	316	24	1189	33	1505	31
Air	686	52	524	15	1210	25
Solid waste	163	12	1191	34	1354	28
Soil and land	123	9	114	3	237	5
Noise	33	2	103	3	135	3
Other	5	0	430	12	435	9
Total expenditure	1326	100	3551	100	4876	100

Source: Based on DETR (1998).

where expenditures rose by over 60 times. A fall in expenditure was noted in two industries: pulp, paper and paper products and machinery and equipment. Overall, however, environmental spending rose by 72 per cent between 1994 and 1997.

The results of the survey should be regarded as approximate, given the relatively high sampling and non-sampling errors that were reported. There are additional reasons to treat these results with caution. For example, it was clear when collating the results that some types of spending, for example, end-of-pipe investments, were easier to identify than others such as clean technology. Also, amortised values for capital spending would be more useful than identifying expenditure in the survey year when processes and installations come on stream. However, this can only be done when information is available for several years.

15.5.3 Household Expenditures

No figures are available on household defensive expenditures in the UK national accounts statistics and details in the literature are very sparse. A major problem is that there is no consensus on what actually constitutes household defensive expenditure; in particular, how they should be delimited. The area is regarded as overly subjective. It is believed that household expenditure is small in comparison to other sectors (industry, government), hence the calculation of these expenditures is less important. It is also argued that household preferences regarding environmental

Table 15.21 Environmental expenditure by industry in the UK, 1997 (million Euro, 1995)

	Current expenditure			Capital expenditure			Total environmental expenditure
	In-house operating expenses	Payments made to others	Total current expenditure	End-of-pipe investment	Integrated processes	Total capital expenditure	
Mining and quarrying	23	34	57	23	0	23	80
Food, beverages and tobacco	114	422	536	91	11	103	639
Textiles and leather products	80	68	148	0	0	0	148
Wood and wood products	34	11	46	34	46	80	126
Pulp and paper products, printing and publishing	137	114	251	126	23	148	399
Solid and nuclear fuels, oil refining	148	23	171	23	0	23	194
Chemicals and man-made fibres	491	285	776	308	103	411	1187
Rubber and plastic products	34	80	114	46	0	46	160
Other non-metallic mineral products	68	68	137	103	91	194	331
Basic metals and metal products	331	183	513	160	11	171	685
Machinery and equipment	34	57	91	11	11	23	114
Electrical and optical equipment	57	46	103	11	0	11	114
Transport equipment	68	80	148	11	11	23	171
Other manufacturing	11	57	68	11	0	11	80
Energy production and water	160	228	388	23	46	68	456
Total expenditure in selected industries	1780	1769	3549	947	377	1324	4872

Source: Based on DETR (1998).

Table 15.22 Comparison of 1994 and 1997 (million Euro, 1995 prices)

Industry	1994	1997	% rise
Mining	57	78	35
Mfr of food and beverages	404	624	55
Mfr of tobacco products	1	15	774
Textiles	124	128	3
Mfr of clothing	13	14	13
Leather and footwear	5	6	22
Timber and wood products	13	123	872
Pulp, paper and paper products	377	216	-43
Publishing and printing	80	178	125
Coke, refined petroleum and nuclear fuels	38	191	6277
Chemicals	624	1189	90
Plastics	107	156	45
Non-metallic mineral, clay, cement	232	335	45
Metals	161	553	245
Fabricated metal products	84	134	58
Machinery and equipment	220	113	-49
Office machinery and computers	3	16	385
Electrical apparatus	24	32	34
Radio, TV and comms	22	45	108
Medical and optical products	15	26	78
Motor vehicles	23	92	309
Other transport equipment	510	83	55
Mfr furniture etc.	17	73	344
Energy production	116	380	226
Water supply	52	74	40
Total	2831	4877	72

Source: Based on DETR (1998).

expenditure will be reflected in industrial production patterns and that these are more amenable to quantitative analysis than household survey-based data.

However, a recent attempt has been made (Simon, 1997) to devise a methodology to link existing qualitative and quantitative data. This was based, in part, on the belief that household defensive expenditure is strongly linked to shifts in lifestyle patterns that are captured well in qualitative data. This reflects the distinction that needs to be made between defensive *expenditures* and defensive *activities*. This can be illustrated by the example of consumers who may choose to grow their own vegetables as opposed to buying organic products or who may cease to buy 'green'

Table 15.23 Merging SERIEE framework classification for defensive expenditure with COICOP groups

COICOP groups	Characteristic activities	Connected products	Adapted products
24 Food and non-alcoholic beverages			
01.1 Food			Biologically cultivated food
02.			
03.			
04.			
05.6.1.1 Cleaning and maintenance products	Charges and prices for the disposal and recycling of electronic wastes and used-up batteries	Noise prevention windows, noise reducing components	CFC-free products, eco-washing machines
....	....	....	....
14.5 Housing			

Note: The omission dots mean that not all rows and columns of the matrix have been represented in the table.

Source: Based on Simon (1997).

products instead of environmentally damaging products and not consume that good or service at all.

The aim of the approach is to merge the classification structure offered by the SERIEE framework (characteristic activities, connected products, adapted products) with the Classification of Consumption per Purpose (COICOP) commonly used in the national accounts. The structure is illustrated in Table 15.23 together with sample entries.

This format allows for the characteristic activities to be organised effectively in an input–output/national accounting framework and helps to relate the characteristic activities to the totals that appear in the section on consumer expenditure in the national accounts. Clearly, such expenditures constitute the upper limits for the estimations of the household defensive expenditure.

To obtain estimates for defensive expenditure associated with the three headings listed above, a literature review of all relevant market surveys was undertaken. This provided figures or percentages that give a first *indication* of the figures found in the national accounts that are of a 'defensive' nature. In many cases a large amount of extrapolation and estimation has been necessary. It is important to note that these estimates are not official statistics. Aggregate figures have been prepared in six broad categories and these are considered in turn. These estimates are included for illustrative purposes only and are not adopted in this final report.

Membership of environmental associations
This is a somewhat indirect form of defensive expenditure and fits under COICOP section 13.8 (environmental protection). The estimates are derived from survey data (Worcester, 1993). Total expenditure is 107 million Euro for 1994–95. ONS (2001) indicates that membership of environmental organisations has risen over the period 1991–99, suggesting that this 1993 estimate is an underestimation of what would now be the case. For example, membership of the National Trust rose by 23 per cent from 2.15 million in 1991 to 2.64 million in 1999 (ONS, 2001). The same is true for most other environmental associations.

Noise abatement
This has been classed under COICOP section 6 (health). Considering only expenditures on double-glazed windows the estimate is 338 million Euro for 1994–95.

Energy efficiency (home improvement)
This considers the purchase of low energy bulbs (£10 per unit) and an average estimate for loft insulation purchase and fitting. The total cost estimate for 1994–95 is 52.9 million Euro.

Green consumerism
This is probably the most debatable category since it is difficult to define in precise terms. The purchase of organically grown food has dual characteristics – the utility derived from the food and the intentional selection of a 'green' product. The National Consumer Council estimate that organically grown products represent 1 per cent of total fruit and vegetable sales. On this basis, total expenditure would amount to 125.5 million Euro for 1994–95.

Taking these four measures only, an estimate of 623.3 million Euro can be derived for expenditures by households in the UK. This is clearly an underestimate of household defensive expenditures, though it is illustrative of using a new methodology and disaggregation of figures, together with the use of qualitative data. This can be seen as a first tentative step towards calculating the defensive expenditure of households under the three categories: environmental protection, restoration and prevention. We do not think that the methodology or data survey techniques are far enough advanced at this time for these estimates to be adopted with sufficient confidence in this report though we believe that the approach outlined above has much to recommend it.

15.5.4 Government Expenditures

Defensive expenditures made by the government are somewhat easier to identify since they will usually be clearly associated with new environmental legislation or political commitments to attain certain quality targets. The first set of results adopt the EUROSTAT classification approach, identifying defensive expenditure by media and type/module. The figures in Table 15.24 are based on those published in the ECOTEC (1993) report. The total estimate for 1990–91 is £4780 million.

The research by Simon (1997) classifies expenditures by sustainability 'themes' and also tries to isolate transfers from the Department of Environment (DoE) to government supported bodies. It is assumed that data are extrapolated from published government reports. Unfortunately, few supplementary details are provided. For example, the entry for 'administration' assumes that defensive expenditure represents 40 per cent of the total administration costs of the DoE – how this figure is obtained is not stated. *The final estimated total is 13 247.5 million Euro which is considered representative for 1995.* Again, this is substantially more than the previously estimated figure but it is very difficult to deduce anything from the change. The indication is that the ECOTEC (1993) study gave a very partial picture of government defensive expenditure in 1990. More recent surveys of environmental expenditures have focused solely on industrial expenditures.

15.5.5 Conclusions

In conclusion, there is considerable uncertainty as to the accuracy and applicability of many of the estimates of defensive expenditures cited above. This degree of uncertainty suggests that it would be inappropriate at this stage to represent the data in this report in any other way than as an illustration of the formative nature of the empirical work on defensive expenditure in the UK at present.

However, recent databases based on the EUROSTAT and OECD surveys show that the degree of accuracy and comprehensiveness have

Table 15.24 Defensive expenditures by UK government

Environmental theme	Million Euro (1995)
Waste management	1 435
Remediation costs	87
Noise protection/abatement	37
Financial transfers to voluntary sector	
DoE[1]	5
ODA[2]	43
MAFF[3]	≈0
Financial transfers to supported bodies	
DoE[1]	2 422
DoE[1] administration	107
Research	42
Environment and international projects	8 706
Countryside and wildlife protection	182
Regional allocations	
Wales	84
Northern Ireland	11
Scotland	112
TOTAL	12 273

Notes:
[1] DoE, later part of DETR; from 2001 part of DEFRA.
[2] ODA now called DFID.
[3] MAFF, part of DEFRA from 2001.

Source: Based on Ecotec (1993) and Simon (1997).

improved significantly in the late 1990s, as Table 15.25 shows. This table reports data either for the year 1999, or alternatively for 1998. It is apparent that data cover environmental protection activities, especially in the field of water treatment, waste management and air protection, rather than expenditure on biodiversity and landscape. Also, data are generally available for the abater principle and work is still needed in most EU countries, in order to come up with robust data for the financing principle.

Original work in the area of environmental expenditure is still needed, but prior to this it is prudent to reach some consensus on what actually constitutes a defensive expenditure or activity before it becomes part of regular official statistics. It is hoped that greater confidence may be attached to future estimates as progress is made in current EUROSTAT and SEEA initiatives in this area.

Table 15.25 Pollution abatement and control expenditure in the EU-15

Country	Public sector		Business sector		Public & private specific Household				Total		Grand total	Exp. per inhab.
	Water, waste, air and other %	Biodiv. and land-scape %	Water, waste, air and other %	Biodiv. and land-scape %	Water, waste, air and other %	Biodiv. and land-scape %	Water, waste, air and other %	Biodiv. and land-scape %	Water, waste, air and other %	Biodiv. and land-scape %	Million Euro	Euro
Austria	4	2	15	0	72		8		98	2	6258.36	778.41
Belgium	30	3	23	0	44		0		97	3	3223.72	318.22
Denmark	42	7	0	0	51		0		93	7	3959.14	759.08
Finland	46	3	36	0	15		0		97	3	899.66	176.45
France	49	2	19	1	23		6		97	3	21650.00	374.88
Germany	31	0	18	0	51		0		100	0	31719.39	389.01
Greece °	55	14	30	0	0		0		86	14	813.07	77.86
Ireland	65	2	33	0	0		0		98	2	430.38	119.63
Italy “	39	0	60	1	0		0		99	1	1775.11	31.00
Luxembourg	97	3	0	0	0		0		97	3	92.32	227.04
Netherlands	54	0	25	0	21		0		100	0	6860.91	444.82
Portugal	54	13	24	1	7		0		86	14	839.27	83.82
Spain *	81	18	1	0	0		0		82	18	3810.82	97.22
Sweden	16	0	41	0	43		0		100	0	1971.78	223.65
United Kingdom	58	6	35	1	0		0		93	7	9306.06	159.08
EU 15 TOT.											93609.99	252.02

Notes:
° Business expenditure of 1996.
“ Business expenditure of 1997.
* Business expenditure of 1993.

Source: OECD Working Paper ENV/OPOC/SE(2003)1

NOTES

1. In this chapter monetary data are expressed in Euro, although they refer to expenditures borne before the entry into force of the Euro as a currency. At the time of the project expenditures were converted into ECU, using the exchange rate valid at that time. Exchange rates used were: £(UK)/Ecu: 1.425; Dfl(NL)/Ecu: 0.451; £(IT)/Ecu: 1.950; DM(Germany)/Ecu:1.96.
2. There are no current studies available in this field.
3. It is the self-declared purpose of this foundation to sponsor only the restoration of monuments damaged by pollution. However, a certain percentage of its expenditures is always attributable to repair work due to general decay.
4. The 1993 figure for the Deutsche Bundesstiftung Umwelt was calculated as the average of expenditures for the years 1991–96.
5. This classification includes both current expenditure (approximately 34–7 per cent) and capital expenditure (approx. 63–6 per cent). Capital expenditure includes: water treatment, soil defence (including hydro-geological measures, waste management, nature-conservation, interventions in areas of high risk and of industrial risk), instruments of environmental policy, environmental information and education, scientific research in the environmental field, creation of employment for environmental protection, management of water resources.
6. For updated data the interested reader can consult the DEFRA (formerly part of DETR) website on http://www.defra.gov.uk.

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PART III

Conclusions and Policy Recommendations

16. Conclusions and policy recommendations

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16.1 INTRODUCTION[1]

This chapter presents the conclusions from each of the principal areas of investigation. These conclusions identify how the results from the analysis can be applied in existing environmental accounting frameworks and point the way for future research initiatives and priorities. From this some recommendations for policy analysis are drawn.

The conclusions on each area of investigation are broadly presented in terms of:

- replicability of the methodology in other European Union countries;[2]
- reliability of the quantitative estimates of the environmental damages in physical and monetary terms;
- consistency of assessment within, and between, countries and over time;
- applications of the methodology and results to specific areas of policy analysis; and
- identified priorities for future work.

The main themes that emerge from this exercise are summarised in Section 16.9.

16.2 REFLECTION ON THE METHODOLOGY: UPDATING DAMAGE ESTIMATES FOR AIR POLLUTION

16.2.1 Replicability

It is necessary to ask whether the methods used here can be applied in countries other than the four selected for this study. The main issue concerns the availability of good quality data through the analytical chain. The situation is summarised in Table 16.1.

The situation across the European Union is set to improve over the next few years with respect to the availability of pollution concentration data, through the implementation of the Framework Directive on Ambient Air Quality (FDAAQ). This requires most member states to increase the number of monitoring stations, measure all pollutants that are the subject of the Directive and to make data readily available to the public, allowing a wider applicability of the GARP methodology using monitored data (concentrations). The most notable gap for stock at risk data concerns material inventories in southern Europe. One can also expect an increased effort in modelling air quality, to ensure that problems are not present in areas where monitoring is absent. Moreover, the use of air modelling is important in its own right as it enables us to determine the origin and cause of damages. Table 16.1 shows in detail that the model developed here is actually replicable in other contexts, as shown in Chapter 9 (see Table 9.22), where estimates of emissions are derived from established emissions coefficients (based on CORINAIR).

To conclude, replicability seems relatively unaffected by issues linked to exposure–response functions and valuation data. Although there are good reasons to suppose that both will vary around Europe for some types of damage, the associated differences are unlikely to be significant compared with other uncertainties in the analysis.

16.2.2 Reliability

Assessment of air pollution damages can now be achieved with a reasonable degree of reliability, *provided that uncertainty is accounted for in a quantified and structured manner*. The caveat appears at first sight to be a contradiction of the main statement. However, all it seeks to do is acknowledge the uncertainties that are known to affect this analysis and that they need to be accounted for throughout the analytical chain.

The reliability of the results generally declines down the impact pathway. Emissions of many pollutants can be quantified to a close degree of accuracy. Dispersion modelling, integration of concentration data with stock at risk, application of exposure–response functions and valuation data then all introduce further uncertainties.

There are problems in interpretation of the results of this work, as evidenced by a tendency to treat all results as equally reliable, which is, in fact, not the case. As a general guide we provide the following reliability ranking,

Table 16.1 Replicability of estimates made, using emissions (modelling)

Stage		Availability of data
Emission inventories		Widely available, though quality varies, particularly for the trace pollutants
Air quality monitoring data and models		Better availability of data in the north of Europe than the south, though models are of course widely applicable
Stock at risk data	Health	Available in all EU-15 countries
	Building materials	Limited availability in the south of Europe, better in the north
	Agriculture	Available in all EU-15 countries considered
	Forests	Available in all EU-15 countries
	Natural ecosystems	Available in all EU-15 countries, though data intensity varies
Exposure–response functions	Health	Sufficient data are available for generalisation of functions to a European average, though there is increasingly good data for country-specific estimates to be made
	Building materials	Available from pan-European UNECE study
	Agriculture	Available, though there are concerns over the use of functions at the pan-European level
	Forests	Little available beyond critical loads and levels, making precise description of impacts impossible
	Natural ecosystems	Little available beyond critical loads and levels, making precise description of impacts impossible
Valuation data		Pan-EU estimates available through the ExternE and Green Accounting projects, though there has been little work done to check on consistency of valuation between EU Member States
Uncertainty data		Uncertainty will vary from country to country, reflecting some of the issues raised in this table

from the most to the least reliable, covering error accumulated through the entire impact pathway:

1. materials damage;
2. acute effects on mortality;
3. crops damage;
4. acute effects on morbidity;
5. chronic effects on mortality;
6. chronic effects on morbidity.

This ranking is derived subjectively but appears reasonably robust, although there are variations around it, depending on the pollutant in question, and the precise impact being assessed. This ranking in decision-making processes has been used in cost-benefit analysis in support of the development of the EU Directive on Ambient Air Quality Limits for Ozone (European Commission, 1998) and the Emission Ceilings Directive.

The question arises of whether the results are reliable enough to be used in the context of national accounts. We believe that the methodology used here is sufficiently transparent. The ease with which data can be updated as new information comes to light is a particularly useful feature of the methodology. Importantly, results expressed in terms of temporal trends are likely to be more robust than absolute estimates of damage. The same would appear to apply to any measure adopted in the context of national accounting.

16.2.3 Consistency

There are three issues here:

- consistency of assessment within countries;
- consistency of assessment between countries;
- consistency of assessment over time.

Of these the last is probably the most important. Without consistency over time it is not possible to see how environmental performance has improved or worsened over the years, and it is this that forms the main rationale for wanting to integrate environmental indicators with national accounts. There are some problems, given that methods for collecting data will change, as will exposure–response functions and valuations. However, the stepwise nature of the impact pathway methodology allows straightforward recalculation of impacts and damages when new data become available.

The main issue affecting consistency of assessment within countries is the availability of data on emissions and air quality. This book shows that for some countries (for example, the UK) data are available throughout the country at a reasonable resolution, whilst in others (for example, Italy) coverage is more patchy. This situation will improve through compliance with the EU's Framework Directive on Ambient Air Quality (FDAAQ).

At the present time problems with consistency of data between countries arise for several reasons:

- use of different instrumentation (particularly important for particle measurement);
- differing densities of air quality monitoring stations;
- assessment of different pollutants.

Again, compliance with the FDAAQ will improve matters in the near future.

16.2.4 Applications

A range of applications where the methodology used here has already been applied have been given in this report. These included analysis of environmental quality objectives such as air quality limits, technical emission standards for emission sources ranging from large combustion plant to vehicles, energy policy strategies and taxation.

The main thrust of this work, however, is the use of impact assessment and monetary valuation as tools for integrating environmental impacts with national accounts. The methodology used provides several types of result that could be set alongside traditional measures of national performance:

- emissions data;
- concentration data;
- receptor (for example, population) weighted exposure indices;
- incidence of pollution-related impacts;
- valuation of these damages.

It is clearly the last of these that is most easily comparable with traditional accounts. However, monetisation cannot be applied to all environmental assets (for example, biodiversity) at least at the present time, and there is thus a risk that some aspects of the natural environment may not be accounted for. If this is a concern, it may be logical to integrate output from another stage of the impact pathway into the national accounts.

Consideration needs to be given to the trade-off between confidence in outputs and the extent to which estimates provide an explicit account of the state of the environment.

Table 16.2 demonstrates that the pathway approach adds very substantial value to the information available. In earlier chapters we compared two different ways of estimating air pollution damages using the impact pathway approach (IPA). We adopted two different methods that yielded different but comparable results. The first one, Approach A, does not adopt the full chain of the IPA (adopting partial IPA) but rather part of it, starting with the second step, that is, concentrations that are interpolated (measured concentrations – impacts – monetary damages). The second, Approach B,

Table 16.2 Application of the pathway approach to developing national monetary accounting estimates of damages

Type of information	Reliability	Usefulness of indicator
Emissions	Generally good	Provides a very approximate indicator of impacts. For example, a unit emission of NOx in London will be much more damaging than the same emission in NW Scotland, though simple emission estimates treat them as equal
Concentration data	Generally good	Useful where thresholds exist, but a weak indicator of impact because it does not account for exposure of sensitive receptors
Exposure indices	Generally good	Allows better targeting of abatement measures towards those sources likely to have the largest impact, but does not provide guidance on whether or not potential impacts are likely to be significant
Physical impact estimates	Variable	Enables conclusions to be drawn as to whether the effects of pollution are likely to be significant and hence worth targeting
Monetary valuations	Variable	Allows direct comparison with traditional economic indicators, using a metric that is universally understood, and reference to issues of most concern to the population

follows the whole IPA (emissions – concentrations – impacts – monetary damages), using emissions calculated with CORINAIR emission factors.

Approach A was used in Chapter 8, whereas Approach B was adopted in Chapter 9. It is therefore interesting to look at the results in terms of monetary damages obtained with both approaches. In the following the analysis concentrates on health impacts as they account for the largest part of the total impacts, and results are available for all countries. However, this comparison is not straightforward for several reasons: some of them are intrinsic to the models used, others depend on the availability of data, the major shortcoming being that Approach B cannot be comprehensively applied for all countries and pollutants. In fact, only Germany includes PM_{10} emissions in the assessment of air pollution impacts, allowing for a comprehensive analysis, whereas for the other countries emission data are only available for a limited number of pollutants considered in this study.

Comparing the synthesis Tables 8.40 (obtained using Approach A) and 9.22 (obtained using Approach B) the reader may have found that there are large differences. Apart from the basic difference between the two different approaches (Approach A uses ground data combined with interpolation, whereas Approach B uses models to calculate concentrations from emissions and ultimately damages), the main sources of differences are as follows:

1. The approaches do not use the same time reference: Approach A uses ground data from 1994–95 whereas Approach B uses emission coefficients taken from CORINAIR 1990.
2. When applying Approach B, PM_{10} local effects were considered only for Germany. Hence damages in other countries are underestimated compared to Germany for this reason.
3. The two methods do not treat import and export of pollutants and eventually damages in the same way: Approach A integrates de facto imports and exports by using concentration data whereas in Approach B these are calculated separately.

There is little that can be done about point (1) above, as this would require an additional project to be undertaken. In terms of point (2) we should just be aware of this fact when comparing across countries. However this issue has been addressed in the follow-up project GREENSENSE where more recent data were made available for most EU countries by TNO (Organization for Applied Scientific Research of the Netherlands). For point (3) we can make an adjustment to make the results more compatible, by accounting both for imports and exports in Approach B (Table 9.22). The adjusted values of health damages are given in Table 16.3 where the adjusted figures should be compared. They show that the two approaches

Table 16.3 Adjusted comparison of health damages with Approaches A and B (million Euro)

	Approach A	Approach B		
	Adjusted	Unadjusted	Net exports	Adjusted
Germany	43 000	50 000	−10 967	39 033
Italy	45 000	23 000	−7 309	15 691
Netherlands	10 000	5 100	+1 183	6 283
United Kingdom	16 000	24 000	−6 677	17 232

Source: See Tables 8.40 and 9.22.

are fairly close for Germany and the UK but rather far apart for the Netherlands and Italy. However, since PM_{10} effects account for a large part of health impacts only German results can be compared. The reasons for these differences need further study but the presumption must be that it is Approach A – the partial IPA – which is the more approximate, that is, the less accurate. As a matter of fact, this also reflects the weaknesses of the monitoring network, especially for some countries, and the limitations of the interpolation exercise using available measured data.

From this section we conclude that, for national accounting, each type of information generated by the impact pathway is useful, though neither of the two described above is ideal at the present time. Ideally valuation should be based on the full impact pathway analysis (Approach B above), so that pollution sources are identified and policy conclusions can be derived. However, technical coefficients and emission data are hardly available on a regular basis. Therefore, a mixed approach could be devised in a future project. In spite of the problems that have been discussed, however, the numbers obtained are fairly reliable and can be useful as inputs into an environmental accounting framework.

16.2.5 Further Work

Further research to reduce uncertainty in exposure–response functions is to be encouraged, though at the present time it does not appear to be a priority for funding bodies for anything other than health. This is unfortunate, because a small amount of research aimed at creating common ground between impact assessment and valuation data could significantly improve the quality of results on crop loss assessment and buildings damage.

Further research on valuation is also needed, especially in the valuation of chronic mortality. At the same time, improving the protocols for the transfer of damage estimates from the site where they were made to another location (benefit transfer) is also needed, in order that the research that is conducted on valuation can be widely applied with confidence.

16.3 DAMAGES FROM AIR POLLUTION

We have already discussed the measurement of air pollution damages from the limited and full impact pathways (Approaches A and B respectively). In this section we consider what has been achieved in terms of damage attribution, a major goal of this study.

16.3.1 Achievements of Present Study in Damage Attribution

In the present study the impact pathway approach was applied to attribute damage costs caused by air pollution to human health, field crops, and material to countries and economic sectors of origin (see Chapter 9). The pollutants included in the analysis were NO_x, SO_2, nitrates, sulphates, O_3 and fine particulate matter. The estimated impacts result from direct exposure of receptors to the pollutant concentrations. As geographically distributed emissions are needed for the analysis, the calculations are based on the CORINAIR 1990 emission database, which is the only database that provides emissions of economic sectors at a high geographical resolution all over Europe. Therefore, the results of the study reflect the environmental situation in 1990.

The damage costs that occur within the EU, caused by all emissions in the EU-15 member states, were estimated at about 130 billion Euro and are summarised in Table 16.4. The assessed damage costs were dominated by health damages caused by fine particles which include sulphate and nitrate aerosols. The precursors of secondary particles are NH_3, NO_x and SO_2. The main contributing sectors analysed in detail in the four countries – Germany, Italy, the Netherlands and the UK – were the energy, road transport and agricultural sectors. Most air pollution related damages occur in a surrounding radius of some 100 kilometres of the emission sources, which means that, especially when considering emissions from small countries, many of the damages from the emitted substances are effected outside the countries. The UK was the only country in the study for which the damage export was smaller than the damage costs caused inside the country by its own emissions. It was also observed that in most countries of the European Union fewer damages are caused by their own emissions than by the emissions of the remaining European

Table 16.4 Attribution of damage costs within the EU in 1990

Source countries	Receptor countries																
	AT	BE	DE	DK	ES	FI	FR	GR	IE	IT	LU	NL	PT	SE	UK	EU	Non EU
	Damage Costs caused by the Source Countries within the Receptor Countries [billion Euro/y]																
EU	2.8	4.5	40.9	2.3	8.9	0.4	21.4	2.0	0.4	15.3	0.1	7.0	1.2	2.1	19.4	128.8	34.9
	Percentage of Damage Costs Caused in the Receptor Countries [%]															[bn ECU/a]	
AT	12.2	0.2	0.9	0.6	0.2	0.9	0.2	0.9	0.1	2.2	0.3	0.2	0.0	0.8	0.1	1.2	1.8
BE	1.1	12.3	3.7	3.8	1.2	1.4	3.7	0.0	1.4	0.7	4.8	13.0	0.4	2.6	1.1	4.4	0.4
DE	47.0	14.2	53.8	43.7	4.8	38.7	13.4	2.0	6.9	15.6	33.3	15.6	0.9	49.6	6.7	34.4	17.0
DK	0.6	1.0	0.9	9.2	0.1	4.6	0.5	0.0	0.6	0.1	0.8	1.0	0.0	8.9	0.8	1.2	0.4
ES	1.7	8.6	3.8	1.8	51.8	0.0	16.3	0.0	16.1	5.6	8.9	6.1	50.4	1.0	7.3	13.5	0.4
FI	0.1	0.1	0.1	0.4	0.0	29.5	0.0	0.0	0.1	0.0	0.0	0.1	0.0	1.9	0.1	0.3	0.1
FR	8.7	33.5	15.3	7.2	16.8	2.1	36.2	0.1	11.2	10.3	31.9	23.8	5.9	5.4	11.4	23.2	2.0
GR	1.1	0.0	0.1	0.2	0.1	0.0	0.1	78.0	0.1	2.3	0.1	0.0	0.0	0.2	0.0	2.1	3.7
IE	0.0	0.3	0.2	0.5	0.4	0.1	0.4	0.0	18.8	0.0	0.2	0.4	0.3	0.2	2.1	0.7	0.0
IT	23.3	3.0	6.9	2.5	8.8	1.6	5.9	18.9	2.8	60.1	4.1	2.4	2.6	2.8	1.2	15.8	6.8
LU	0.2	0.3	0.4	0.2	0.1	0.1	0.2	0.0	0.1	0.1	1.6	0.4	0.0	0.1	0.0	0.3	0.0
NL	1.2	8.0	4.6	7.1	1.0	2.3	3.8	0.0	1.4	0.6	5.6	13.8	0.3	5.0	1.8	4.9	0.5
PT	0.0	0.6	0.2	0.1	6.8	0.0	1.1	0.0	2.4	0.1	0.6	0.4	36.0	0.0	0.7	1.6	0.0
SE	0.2	0.3	0.4	1.5	0.0	11.7	0.1	0.0	0.2	0.0	0.2	0.3	0.0	8.1	0.3	0.5	0.3
UK	2.6	17.5	8.6	21.2	8.0	7.0	18.1	0.0	37.8	2.0	7.7	22.4	3.3	13.4	66.5	24.7	1.2

countries. By comparing imports and exports of damage costs within the European Union, net exporting and net importing countries were identified. The net exporters within the EU in 1990 were Spain, France, Greece, Ireland, Italy, Luxembourg, Portugal and the UK. These results show the highly regional character of environmental problems related to air pollution and the importance of the imports and exports of damages for their attribution to sources.

The analysis of damage attribution to the CORINAIR main sectors of the four countries Germany, Italy, the Netherlands and the UK has been reported in physical as well as monetary terms (see Chapters 8 and 9). This allows the policy maker to focus on the physical impacts and ignore the monetary evaluation if the latter is felt to be too uncertain. Uncertainties in the functions used and the use of alternative exposure–response functions as well as the effects of further pollutants are discussed in the sensitivity analysis carried out in detail for the UK (Section 8.4). As noted there, the impacts of the air pollutants on complex ecosystems, for which no reliable exposure–response relations are available, are not included in the analysis of damage attribution. One way of including these effects would be by reporting exceedances of thresholds and critical levels/loads by using the concentration maps estimated as interim results in the impact pathway analysis together with maps which show ecosystems' sensitivities towards exposure to different pollutants. Much effort has to be invested, however, in the development of exposure–response relations in this area before such an approach can be implemented.

16.3.2 Conclusions and Outlook

One aim of the study of damage attribution was the description of interactions between the economy and the environment. This is especially important in the area of air pollution, because the location of the emission sources and the meteorology highly influence where the impacts will occur. In the study a methodology for the estimation of geographically distributed damages caused by individual economic sectors was developed. The damages estimated from 1990 emissions using the methodology show the interactions between economic activities and the environment and the geographical distribution of damages due to different sources. These results can therefore be used for a detailed analysis of the international problems concerning air pollution in 1990. Regions in which high damages occur and the damages related to emission sources can be identified. This allows us to derive geographically differentiated aims for emission reductions within the European Union. Furthermore, the impact pathway analysis can be applied to estimate exceedances of thresholds or critical levels/loads, which can be set according to political decisions or scientific knowledge. On this basis effective strategies of emission reductions and avoidance costs can be estimated by using economic models. The analysis of year on year changes in the results of environmental impact assessments can additionally show the effectiveness of political measures in the area of air pollution. These can be used to control and further improve the environmental measures. The environmental indicators which are currently provided in environmental accounting systems include neither a detailed geographical reference nor environmental impacts connected to emissions and therefore cannot be applied in a detailed analysis of interactions between emissions of economic sectors and environmental impacts or threshold exceedances in the area of air pollution.

CORINAIR 1990 was used as the emission database in the study, because it is the only available database which provides emissions of economic sectors in a high geographical resolution. Therefore, the nomenclature for economic sectors used in the study corresponds to the technical nomenclature of CORINAIR 1990. For a direct linkage of the results to the economic data in national accounts and input–output tables, it would be necessary to estimate impacts and damage costs in the general industrial classification of economic activities within the European Communities (NACE). This classification is used by Eurostat and is increasingly being adopted by individual national statistical offices for their national and environmental accounting systems. One way of using the NACE nomenclature is to translate the results estimated for CORINAIR SNAP codes to NACE. However, this procedure requires the subdivision of results estimated for individual SNAP sectors and their distribution to several NACE sectors, especially for the energy and transport sectors. This is not ideal, because the transformation can only be an approximation and the high geographical resolution of the CORINAIR emissions has to be left out. A better option is to use the basic data which were collected to build the CORINAIR database and to assign these data according to NACE nomenclature. However, it was not possible to estimate the damage figures presented in this study according to the NACE nomenclature. This is an important issue for further investigation.

The study of the attribution of damages to source sectors and countries has shown that, especially where air pollution is concerned, a European environmental accounting system including the estimation of transboundary damages is very important in order to track environmental developments within the EU. This system also serves to highlight the effectiveness of measures as well as indicating in which area of European Union policy actions should be initiated to reduce environmental problems.

16.4 DAMAGES TO WATER

The GARP II project provided the opportunity to make a first assessment of the existing data on water damages and their potential use in environmental accounting. The study has identified alternative measures of water quality as well as the sources of valuation that could be adopted. Three different water uses have been considered (that is, as a sink for industrial emissions, recreation and amenity), and the potential for monetising water quality for each one investigated. It is clear that the development of appropriate methodologies remains a priority before damage values can be generated with sufficient credibility to guide policy.

Because of the preliminary nature of this work it has not been possible to gauge the potential scale of the damages to water on a national basis. It is evident that this would require a broader coverage of the likely damage categories than attempted here, together with a more established methodology and database of values. Notwithstanding this, we strongly believe that in reality the annual damages in the four countries involved in the GARP project, as well as in most other European countries, will have significant policy implications.

16.4.1 Replicability

The potential for water quality measurement is good for waterways since data are collected routinely for most large river catchment areas. As noted in this study, however, the measure adopted at a given site generally reflects the priority identified for that area and elements considered at risk. Consequently, there exist a range of measures that identify different aspects of water quality which are currently used in different locations within different countries. It is therefore uncertain as to whether it is currently possible to use a single measure of water quality throughout Europe.

It is only the lack of similar monitoring data types that prevents the methodologies adopted in our study from being applied in other European countries. The practical constraints on their use, however, relate also to the availability of monetary data. In particular, the cost data on water treatment required for one methodology have been very difficult to identify, because of the inadequacy of existing accounting conventions and the practice of commercial confidentiality.

These factors would suggest that comparison of water damages across countries in Europe is not likely to be possible in the immediate future. In order to enable comparison, further work needs to establish conventions in physical and monetary measures that encompass the broadest range of impacts. Fortunately, the Water Framework Directive, which mandates the eventual full cost pricing of water, has given an impetus to studies of water damages, which will greatly facilitate accounting work in this area.

16.4.2 Reliability and Consistency

There is reasonable confidence in the reliability of the water quality data used in the study. Care has been taken to make base measurements under a wide range of flow conditions across the year in order to account for seasonal variations. The measurements themselves have also been accepted as being sufficiently rigorous. One of the main problems in the use of water quality data, however, is that the average physical value adopted is extrapolated across a wide geographical area, often with a correspondingly wide ecological variation. Clearly, the validity and reliability of this extrapolation can only be tested with a comprehensive measurement programme.

The monetary valuation data included in the study relate only to recreational and amenity uses (that is, they exclude commercial uses). These are derived from studies that used contingent valuation and hedonic techniques. We are reasonably satisfied that these values can be validly transferred to the present research context. However, reliability issues remain in terms of the linkage between pollutant loads and fish stocks, and particularly the species involved, as this determines, to a large extent, the willingness to pay for recreational angling.

16.4.3 Applications

We consider that the measures of water quality identified are sufficiently developed to form the basis of an environmental accounting framework for water. This belief is supported by the substantial progress made in the development of physical water accounts in the UK, France, the Netherlands and Spain.[3] Our main concern is with the paucity of monetary data that can reliably be applied to the physical data. At present, this is insufficient to give more than a partial picture of total damage costs. Further development of the accounting requirements relating to water treatment costs are therefore required before we can recommend a wider application of water damage accounting.

16.4.4 Further Work

The development of an impact pathway approach to water damages is very complex and was basically not possible within the framework of this EU project. The wide variation of methodologies used in water damages would make this type of analysis impractical. However, there is ongoing work on sources of water pollutant loads (more so as the programme initiated under the Water Framework Directive unfolds) and over time this could be generalised and extended across countries to help policy design.

In order to apply the water treatment cost methodology, the construction of a database of water treatment costs is urgently needed. It would then be possible to monetise the value of industrial and agricultural uses. A further priority is a formally agreed list of water damage categories that encompass all uses of water. From this, it would be possible to identify the areas where further development and application of monetary valuation techniques need to be concentrated.

It is envisaged that future efforts in these aspects of water damage estimation will facilitate the inclusion of the water sector in further iterations of environmental accounting procedures.

16.5 DAMAGES TO LAND

Significant progress has been made in assessing the costs of contaminated land within this phase of the project. The study has assessed the state of available databases of contaminated land, evaluated the application of available valuation data and made an initial quantification of expenditure on land remediation. Nonetheless, the approach is still at an early stage and considerable further effort is needed to derive a workable methodology for assessing the flow of damages for use within national environmental accounting.

The study has highlighted how important an issue contaminated land might be. The Caracas Research Programme estimates that there are around 750 000 sites of contaminated land within Europe. Given that the remediation liability for a country such as the UK on its own is estimated to be 30 billion Euro, the extent of land contamination at the European level is such that the potential remedial costs required are likely to be enormous.

16.5.1 Replicability

The analysis in this book has estimated the current stock of contaminated land and, for some countries, made estimates of the likely remediation liability that this represents. There are considerable problems in producing such estimates for other countries, because of problems with data availability and standardisation of contaminated land registers. Registers are related to the national definition of a contaminated site. In conjunction with a priority setting system, registers are generally regarded as a means of arriving at a consistent, effective and fair allocation of resources for solving contaminated land problems. However, consistency in definition varies across the EU and therefore a comparison between registers can only be tentative. Even within the four countries assessed here (the UK, Italy, Germany and the Netherlands) there are widely varying definitions and data availability.

It is likely that these problems may be more acute in other European countries, as the Netherlands and Germany (along with Denmark) have been identified as having the most advanced systems for national contaminated land registers.

Similarly, because of variation of legislation by country, any remediation cost estimates are country specific, due to the nature of the clean-up required in different countries. This means that any approach to cost assessments, at present, has to be country by country.

Overall, considerable further work is required at a European level (through standardisation procedures, remediation requirements, and so on) in order to allow any kind of comparison between countries. There are European programmes in place which will hopefully look at such issues, and it is likely that the situation will improve in the future.

16.5.2 Reliability and Consistency

The assessment contained here only looks at the stock of contaminated land. It does not attempt to assess the environmental damage to land as the basis of national environmental accounting. Nevertheless, even within the analysis of the stock, there are very serious problems in terms of reliability. These mostly centre on data availability and consistency.

The reliability of the initial cost estimates presented are difficult to judge. Certainly there is a high level of uncertainty because of the variability between individual sites. As far as we can ascertain from historical cost data, the values presented should be of the correct order of magnitude.

16.5.3 Applications

The information collected in this book provides a useful basis for estimating the remediation costs of contaminated land. This information would be useful for national accounting, showing the degree of land contamination and providing information on land remediation expenditure.

For some countries, such as the UK, new legislation and registers should allow greater accuracy, both in the definition of the scale of the problem, and in developing accurate cost data for different types of sites. Once these registers are up and running, data should emerge on the annual numbers of sites added to the registers or removed from them following remediation, and thus some idea of the flow of environmental damage.

16.5.4 Further Work

At present there is wide variation concerning data availability in different EU countries. The fact that there is no standard classification means it is difficult to compare information and costs between countries. There are some European programmes in place but it is unlikely that procedures for identification and classification of contaminated land will be standardised across Europe in the near future. Owing to the variance between countries, in the definition of contaminated land and hence format of national registers (where held), some form of comparative methodology would need to be evaluated.

Similarly, the remediation costs in different countries vary widely according to the legislation in place. Improvements in costs of remediation could be achieved by using estimates per site type. This is dependent on the registers including information on past industrial use, or contaminants present. Nevertheless, because of the differences in legislation, such an approach would still need to be country specific. Further work is needed to improve costs of remediation by using estimates per site type, based on historical use, along with improvements in the costs of remediation for special sites with the use of European rather than US data.

Given the site specificity of contaminated land and potential effects, it seems unlikely that an impact pathway approach quantifying effects on human health, crops and so on, is possible. Instead the analysis here has looked at remediation costs. However, the assessment within this study has only quantified the stock of contaminated land. What is needed as the basis of environmental accounting is the *change* in contaminated land from anthropogenic activity on an annual basis (that is, the flow of damage). It is possible to estimate what the remediation expenditure per year would be for a number of the countries studied, within a given timeframe (for example, by assuming the target for remediation of all current sites is 2010). However, these remediation costs will be for remediating land contaminated as a result of historical contamination, not current contamination. Therefore, further work is needed to look at the levels of current-day land contamination, which, though lower, do continue through activities and accidental releases.

Finally, there are important issues of cost internalisation for contaminated land, in terms of whether lower values of land are already reflected in the present accounts. It has not been possible to look at this issue within the current study, though this is highlighted as an area requiring further research in the future.

16.6 FOREST AND ECOSYSTEM DAMAGES

16.6.1 Issues Raised

From a monetary valuation perspective, the development of a comprehensive and valid common methodology for forest and ecosystem damage is a demanding exercise. A complete valuation of all functions of nature is even considered impossible according to some authors like Geisendorf et al. (1996). They claim that it is not feasible to show a value of biodiversity in any comprehensive sense in environmental accounting, the main reason being that the 'primary value' of nature cannot be monetised (Gren et al., 1994).

However, by means of CVM studies subjective monetary values can be estimated, at least for parts of natural systems. A variety of studies in this area are now available – in particular on ecosystem damage. Since ecosystems are so complex and so individually different, results cannot easily be transferred or applied to situations other than the original situation for which the methodology was developed and tested. In spite of these observations, however, it is interesting to note that the method has produced internationally converging results when comparable research questions are asked.

Recently, several studies on the estimation of recreational benefits of forests have been undertaken. Unfortunately, none of them is directly concerned with forest damage, for example, the consequences of forest damage on recreational value. This is an area for further research.

Bearing these difficulties in mind, an alternative approach has been developed in which the restoration costs of a politically agreed minimum level of biodiversity in Germany and the Netherlands are taken. Taking the restoration costs to achieve collectively decided objectives is, of course, a completely different approach from the assessment of environmental damage costs through estimating individuals' willingness to pay to avoid this damage. It is also noted that further, more detailed research on actual net restoration costs is required, which takes the various agricultural transfer payments into account.

Taking restoration costs as an estimate of damage begs the question of whether the restoration is justified – that is, do the benefits exceed the costs? Of course, from the perspective of sustainable development the current state of knowledge about restoration costs is important because they will determine the eventual programme for the implementation of a sustainable development strategy. But to assume that, if a society has agreed on a certain level of restoration, it must have carried out the relevant comparison of costs and benefits is to assume that the exercises of environmental accounting, such as we are undertaking, have already had their full impact, and clearly that is not the case! Hence, while politically agreed estimates of restoration are important, and useful as a guide to the social costs of the pollution, they cannot be seen as the final word on this topic.

16.7 CROPS

The measurement of economic impacts of air quality improvements on the agricultural sector is best carried out with a sectoral model which makes use of well-defined supply and demand functions for all agricultural inputs and outputs. These are empirically estimated at the farm level and aggregated at regional or national level. By equating aggregate supply and demand market prices are determined. Until now such a model has not been available and all estimates of economic impacts of improved air quality will have been biased as a consequence of ignoring supply and demand movements.

In the analysis undertaken in Chapter 10 the Dutch Regional Agricultural Model (DRAM) is used. This model assumes the existence of regional farms, which compete for fixed resources to optimise national gross margins from agriculture. There is some aggregation bias that results from grouping individual farms with differentiating features into one regional farm and by grouping individual components into the model's activities. Much of the heterogeneity of farms and their components is lost in the aggregation, though the severity of the bias depends on the magnitude of the identified differences. There are other issues resulting from the use of a model that reduces the accuracy of the results, namely that quality changes are not taken into account and that since DRAM is a comparative static model, the time-path towards completion of the adjustments is unknown.

Notwithstanding these shortcomings we believe that the damage calculations presented here contribute to the development of better estimates of agricultural benefits of air quality control. From an economic surplus or economic welfare point of view, reducing ozone concentration levels is important. However, economic benefits are unequally distributed between activities, regions and hence also between individual producers. The DRAM model estimates a total economic damage of 327 Euro million in 1994 because of ozone pollution to crops. Damages to crops caused by SO_2 seem to be negligible.

A comparison with the simple multiplication method (yield changes valued at fixed prices) reveals that economic adjustments will result in farmers benefiting less from ozone control and consumers benefiting more. Whether total economic surplus is under- or overestimated by the simple multiplication method is an empirical question. In our analysis the difference in magnitude of total economic surplus as calculated with a full economic model and as calculated with the simple multiplication model is relatively modest. This confirms earlier findings.

16.8 DEFENSIVE EXPENDITURES

Data on defensive expenditures[4] (that is, expenditures undertaken to reduce the impacts of environmental pollution caused by anthropogenic activity) are an important component of any accounting system. A great deal of work has been going on in this area for some time, but there is still some lack of consistency of approach across countries, and much remains to be done.

Within the EU, Eurostat is taking the lead in addressing these issues, through the SERIEE (Système Européen pour le Rassemblement des Informations Economiques sur l'Environnement) methodology for reporting such expenditures. This approach has been taken on board during the SEEA revision and is now incorporated into the updated SEEA (Integrated System of Environmental and Economic Accounting). The Eurostat approach only considers expenditures linked to activities whose main objective aims at preventing, reducing and eliminating pollution or at protecting the natural environment. This implies that activities such as water supply, saving of energy or raw materials are not included in the definition of environmental protection – even if beneficial to the environment – since their end-purpose is not the protection of the environment. One could argue, with some justification, that these expenditures should be integrated into the full natural resource use and management accounts, and eventually this will happen, but in the short to medium term its objective is to get consistency in the more limited categories of environmental expenditure.

The Eurostat approach is also conceptually distinct from an alternative approach in which, instead of looking at actual expenditure, one estimates the expenditure *required* to achieve a given level of environmental quality and a given efficiency in the use of natural resources. This latter approach has been used in the ISPE classification by the Italian government and by Dutch environmental statistics. For similar reasons to those discussed in the section on forestry, we are sympathetic to this kind of approach as a means of measuring environmental costs, but we do not believe it is a substitute for the estimation of actual expenditures undertaken in response to actual levels of environmental pressure.

The SERIEE approach presents data from three perspectives: on the basis

Table 16.5 Comparative table of official statistics available by institutional units

Country	Government	Industry	Households
D	Attained	Attained	Partially attained
I	Partially attained	Partially attained	Not attained
NL	Attained	Attained	Attained
UK	Not attained	Partially attained	Not attained

of where the environmental item was produced, where the commodity or service was used and who has financed the activity. It also divides expenditures by environmental media: ambient air and climate, waste water management, waste management, soil and groundwater protection, noise, biodiversity and landscape, radiation, research and 'other'. Presently, EU member states are struggling to adapt to the SERIEE approach. Table 16.5 summarises the status of the four countries in this study with respect to reporting expenditures by institutional unit of government, industry and household taken from Eurostat official statistics. Tables 16.5 and 16.6 report whether the present situation is one where the SERIEE methodology has been 'attained', 'almost attained', 'partially attained' or 'not attained' in respect of the different sectors and environmental media (for all four countries), according to the information available in the mid-1990s.

Both tables show: (a) a particular lack of conformity with respect to households and (b) major gaps for biodiversity and landscape, radiation and some gaps for all classes.

As a result of the inconsistencies highlighted above, it is difficult to compare data on defensive expenditures across countries using the data available at the time for the years 1994 or 1995. However, based on a major effort by Eurostat, and broadly following its methodology, a synthesis of statistical data has been collected for government and business only and is reported in Table 16.7 from 1990 to 1997. The table excludes expenditure on biodiversity and landscape protection. The table shows:

(a) The most complete data for business and government expenditures are for Germany and the Netherlands.
(b) The amount of expenditure for these two institutional sectors has been shifting from investment expenditure to current expenditure over the period 1990–97.
(c) In total, expenditure on these two items is around 1 per cent of GDP, which is considerably less than the estimated damages (see Chapters 8 and 9). Due to lack of coverage of some media (for

Table 16.6 Comparative table of official statistics available by domain

Country	Ambient air and climate	Waste water management	Waste management	Soil and ground-water	Noise and vibration	Biodiversity and landscape	Radiation	Research development	Other activities
D	AA	A	A	AA	AA	NA	NA	PA	PA
I	PA	PA	PA	NA	NA	NA	NR	PA	PA
NL	A	A	A	PA	PA	PA	NA	PA	PA
UK	PA	PA	PA	NA	PA	NA	NA	PA	PA

Notes: A: attained; PA: partially attained; NA: not attained; AA: almost attained; NR: not relevant.

Table 16.7 Investment and current expenditures in project countries

	Investment Expenditures (million Euro)			Current Expenditures (million Euro)		
	1990	1995	1997	1990	1995	1997
Germany						
Industry	3 060	2 695	1 807	4 590	6 650	6 053
Government	5 058	6 394	4 251	4 551	8 251	6 969
Total	8 119	9 089	6 058	9 142	14 901	13 022
As % of GDP		*0.5%*	*0.3%*		*0.8%*	*0.7%*
Italy						
Industry	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.
Government	308	n.a.	n.a.	70	n.a.	n.a.
Total	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.
Netherlands						
Industry	869	679	1 040	823	964	1 024
Government	433	739	589	1 696	3 441	3 826
Total	1 302	1 419	1 630	2 519	4 405	4 850
As % of GDP		*0.5%*	*0.5%*		*1.5%*	*1.6%*
UK*						
Industry	1 177	1 550	1 678	n.a.	1 467	4 495
Government	807	n.a.	n.a.	2 330	n.a.	n.a.
Total	1 984	n.a.	n.a.	n.a.	n.a.	n.a.

Notes:
Up to 31 December 1998 Eurostat data are published in ECU and here are reported in Euro using the conversion factor 1:1.
* UK data are for 1994 (and not 1995).

Source: Eurostat, New Cronos database.

example, biodiversity) and some categories (for example, households), these figures are clearly underestimates of total defensive expenditure.

(d) Data are missing for Italy for both government and industry. Government data are for central government only, which, from survey data, are only about one-sixth of total public spending. However, the situation has improved in recent years, as a substantial effort has been made to calculate public environmental expenditure both at central and regional level. For industry no data are reported in a manner consistent with the Eurostat methodology. In the case of the UK, historical data are now available for both industry and government but yearly coverage is patchy.

Table 16.8 Synthetic table of statistics and estimates of environmental expenditure in 1994 (million Euro)

Country	Government		Industry		Households	
	Official statistics	Surveys/ estimates	Official statistics	Surveys/ estimates	Official statistics	Surveys/ estimates
D	14 422		15 875		1350	
I[1]	1195°	6199*	n.a.	512°	n.a.	3584*
NL[2]	4300		3900°		200°	
UK[3]	n.a.		n.a.	3335*	n.a.	7463*

Notes:
Exchange rates: £(UK)/Ecu: 1.425; Dfl(NL)/Ecu: 0.451; £(IT)/Ecu: 1.950; DM(Germany)/Ecu:1.96.
* Data characterised by high uncertainty (see below for detailed explanations relating to the country concerned).
° Partial data (see below for detailed explanations for the country concerned).
n.a. Not available (used only for official statistics).
1 Official statistics for the public sector only consider central government, whereas estimates include regional and local authorities. Most recent data for the industrial sector refer to 1992.
2 Reported data for the Netherlands do not give environmental expenditure, but the environmental costs and refer to the year 1995.
3 In the UK, there is a lack of official statistics.

Source: Various sources (cf. Chapter 15).

Chapter 15 reported data from different sources, including some survey estimates. These are summarised in Table 16.8 and provide some comparative data, including household expenditure, where possible. Note that the statistics in this table are *not* consistently based on the Eurostat methodology. That is certainly the case for the Netherlands, whose national system is based on costs of attaining certain environmental standards, and not on the actual expenditures undertaken. (For Table 16.5, however, the Netherlands provided data on a Eurostat-consistent basis.) This table shows:

(a) Household expenditure when estimated on the Eurostat basis can be substantial; in the case of Germany it is only 1.4 billion Euro, but for Italy and the UK, the figures are 3.6 billion and 7.5 billion respectively (based on surveys).
(b) Official government and industry estimates have many 'holes' in them. In Italy official government spending was only 1.2 billion Euro, but surveys which included regional spending came up with Euro 6.2 billion.

(c) The data in Tables 16.5 and 16.6 may not match, because of differences in coverage.

In summary, this overview of European official defensive expenditure statistics calls for more consistency and effort in classifying expenditure. It is time-consuming for statistical offices to work with a multitude of classification systems. In fact, recently Eurostat and OECD have been working together on common data collection. A unique system is in the process of being agreed at international level, in order to improve comparability among countries. It is likely that in the future a harmonised approach will be used at international level thanks to the implementation of the revised SEEA. However, this is not expected to be the last revision of SEEA, as some consensus is still required for certain areas.

16.9 OVERALL POLICY CONCLUSIONS AND RECOMMENDATIONS

The purpose of this exercise was to investigate the scope and applicability of the impact pathway and damage cost methodology to the development of environmental accounting. The basic results from this study are summarised in Tables 16.2, 16.4, 16.7 and 16.9 of this chapter. Table 16.9 shows calculations obtained using the partial impact pathway analysis (Approach A, cf. Section 16.2.4). They present a picture where monetised environmental damages in 1990 were between 2 and 4 per cent of GDP, or between Euro 306 and 778 per inhabitant.[5] In order to obtain data for more recent years we would have to repeat the exercise as data become available. This can now be done relatively easily. As in previous studies the great majority of environmental damages are to health, with premature mortality and morbidity values being about equal. The attribution analysis by country (Table 16.4) shows that there is a great deal of transfer of damages across borders. The main contributing sectors in the four countries were energy, road transport and agriculture. Interestingly, most EU countries suffer more from damages caused by the emissions of other European countries than from their own emissions (the UK is an exception). The analysis also identified net exporting and net importing countries.

The data on defensive expenditures paint a patchy picture and show that much remains to be done in this area. From what one can see, expenditures are of the order of 1 per cent of GDP, but there are many gaps. This is clearly an area where further work is needed.

Our conclusions on the main areas identified at the beginning of this chapter are given below.

Table 16.9 Damage costs caused by the pollutants SO_2, PM_{10} and O_3 in Germany, Italy, the Netherlands and the UK

	Germany	Italy[a]	Netherlands[b]	United Kingdom[c]
	Damage costs (million ECU[d]/a) unless otherwise stated: base year 1990			
Human health				
Mortality	22 191	22 564	5 084	7 952
Morbidity	21 157	21 936	4 926	7 932
Subtotal	43 348	44 500	10 010	15 884
Percentage of GDP[e] (%)	2.73%	4.41%	1.9%	1.75%
Costs per inhabitant (ECU/(person*a))	532	778	651	272
Crops				
Subtotal	1 611	2.2	154	754
Percentage of GDP (%)	0.10%	2 e-4%	0.06%	0.08%
Costs per inhabitant (ECU/(person*a))	20	4e-2	10	13
Material				
Subtotal	136	N.A.	10	1 250
Percentage of GDP (%)	0.01%	N.A.	3.8e-3%	0.14%
Costs per inhabitant (ECU/(person*a))	2	N.A.	1	21
Total	45 094	44 502	10 174	17 888
Percentage of GDP (%)	2.8%	4.4%	3.9%	2.0%
Costs per inhabitant (ECU/(person*a))	554	778	662	306

Notes: The background levels were assumed to be PM_{10}: 10 μg/m^3, O_3: 20 ppbV (AOT40 crops: 0 ppbVh), SO_2: 1ppbV.

[a] Results for Italy do not include damages due to O_3 or material impacts.

[b] Results for the Netherlands include morbidity impacts of CO at 1.8 million ECU per annum (assumed background for CO: 0.15 ppbV). Wet acid deposition was assumed to have an average value of 100 meq/m^{2}a.

[c] Results for the UK include material damages due to acidity.

[d] In 1995 prices.

[e] European Commission (1997); Eurostat (1997).

16.9.1 Replicability

We find that the availability of good quality data is the main limiting factor in considering how far the methods used in this book can be replicated in the wider European context. In particular, it is apparent that monitoring the physical environment and making an inventory of stock at risk is less comprehensive in southern European countries. Within this context, the damages from air pollution are at present estimated using the most complete analytical chain, reflecting the greater attention that has been given to air pollution modelling in both Markandya and Pavan (1999) and in this work. Water and land-based pollution have been examined for the first time in this volume. Methodologies for damage cost estimation and attribution are therefore less advanced for these media. It has also proved to be more problematic to apply the impact pathway methodology to them as a result of the greater complexity identified as being inherent in the pollution impacts in these contexts. Nevertheless, some progress has been made and useful results obtained.

16.9.2 Reliability

The more complete impact pathway developed for air pollution allows us to identify explicitly the sources and extent of uncertainties which, in turn, determine the reliability of the damage cost estimates in an environmental accounting context. It is unavoidable that, with each step along the analytical chain, uncertainty is added. It is therefore clear that uncertainty is likely be significant in the final estimates. However, we think that as long as the degree of uncertainty is made explicit throughout the damage estimation procedure this transparency will allow the results to be helpful to policy makers.

For the reasons outlined above, the studies of water and land damages do not currently facilitate identification of uncertainties throughout the impact pathway. In both cases the lack of standardised procedures for estimating physical damages, combined with incomplete data records, have therefore reduced the reliability of estimated results. Clearly, these are first attempts at quantifying damages to these media and with further research effort in environmental monitoring and appropriate methodologies the reliability of these results could be improved substantially.

16.9.3 Applications

The GARP exercise has developed a number of methodologies that can and do help decision makers to be more confident that policy initiatives will be targeted on the basis of pollutant attribution and in correct measure. This is most advanced in the case of air pollution where the modelling work undertaken has had wide application in the design of air quality strategy. The results obtained for water and land, whilst being preliminary, also provide a useful indication as to how priorities should develop in these areas.

Applications to environmental accounting are more straightforward than in many other policy contexts for air and water due to the fact that temporal changes in environmental impacts are likely to have greater robustness than absolute damages. This is partly because willingness to pay valuations and physical measures of environmental impact are, at present, most easily defined over more incremental-type changes whereas absolute damages from anthropogenic activity are difficult to isolate accurately. We are confident that in the case of air pollution it should be possible to establish a core European monitoring team to undertake periodic, perhaps annual, reporting of damage costs in the form of satellite accounts. This could come into effect immediately.

Methodologies for the inclusion of water and land in an environmental accounting framework are not sufficiently developed at this stage to allow us to recommend the adoption of satellite accounts at this point. However, we think that in both cases there is a necessity for standardisation and centralisation of databases that would allow reporting procedures to be established. Further research will ensure that methodological developments could parallel and inform this data gathering exercise. In turn, this should allow the formation of satellite accounts.

The GARP exercise has attempted to apply welfare-theoretic willingness to pay valuation measures to the physical environmental impacts in order to present damages in monetised terms. We believe that, where possible, this is the most appropriate and meaningful way to express damage costs. It is clear, however, that where damages are non-marginal or very complex in their effect, avoidance costs can be used as a broad proxy for willingness to pay based valuation. Avoidance costs are, of course, useful to policy makers in their own right. In the context of environmental accounting, however, we think that these costs should only be used as a monetary indicator where willingness to pay measures are clearly invalid. We therefore suggest that future developments in environmental accounting should recognise the strengths and weaknesses of these two monetary indicators and adopt them according to the context, as highlighted. Finally we recognise that not all impacts have been valued in this approach and that any monetary valuation will have to be accompanied by a reporting of physical damages, especially in those areas where monetisation is weak or controversial.

NOTES

1. In this chapter monetary data are expressed in Euro, although they refer to expenditures borne before the entry into force of the Euro as a currency. At the time of the project expenditures were converted into ECU using the exchange rate valid at that time. Exchange rates used: £(UK)/Ecu: 1.425; Dfl(NL)/Ecu: 0.451; £(IT)/Ecu: 1.950; DM(Germany)/Ecu:1.96.
2. Reference to the EU here means the 15 member states prior to the May 2004 accessions.
3. See Vaze (1998) for a review of the material.
4. Defensive expenditures are also referred to in the statistical literature as environmental expenditures.
5. We also report in the book the physical damages corresponding to these impacts, given the uncertainties surrounding monetary valuation.

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