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Valuing the benefits of environmental policy: The Netherlands

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Abstract

This study seeks to set priorities for environmental policy in the Netherlands. The report focuses on seven environmental issues including: climate change, acidification, low level ozone, particulate matter, noise, eutrophication and land contamination. These issues are prioritised using three different approaches: damage assessment, public opinion and 'disability adjusted life years'(DALYs).

The damage assessment approach largely follows that of the European Commission DG Environment study 'European Environmental Priorities: an integrated economic and environmental assessment' (RIVM et al, forthcoming 2001). It is based on a logical stepwise progression through emission, change in exposure, quantification of impacts using exposure-response functions, to valuation based on willingness to pay. The existence of significant uncertainty in assessment of environmental damage is dealt with by conducting a transparent sensitivity analysis for each issue, this demonstrates the consequences of uncertainty on the robustness of our conclusions. The public opinion approach makes use of European and national surveys to determine the importance of environmental issues as perceived by the population of the Netherlands. The DALY methodology largely follows that of Murray and Lopez (1996). This procedure combines years of life lost and years lived with disease or disability that are weighted according to severity.

According to the damage assessment approach the priorities, in terms of potential benefits from full control, are low level ozone, land contamination and particulate matter, followed by acidification and climate change, whilst noise and eutrophication are estimated to yield the lowest potential benefits from control. However, in the absence of cost estimates no conclusions can be reached on the desirability of control measures. Public opinion surveys show that environmental issues other than the seven considered in this study are a major concern for the Dutch public, namely chemical release and oil pollution. However, focusing on the seven issues considered in this study, the Dutch public rank, climate change, acidification, eutrophication and air pollution from cars (interpreted as low-level ozone and PM10) as the issues of most concern. According to the DALYs approach the health effects of air pollution from particulate matter, and to a certain degree from low level ozone, dominate the disease burden. The future disease burden is largely due to changes in the population structure, i.e. an increasing, aged population. Another environmental problem associated with a high disease burden is noise exposure from road and air traffic.

Based on a simple 'Borda count', a final ranking for the environmental issues is made. This study concludes that land contamination, climate change and particulate matter are top priority environmental issues in the Netherlands, followed by acidification, low level ozone, eutrophication and finally noise. These findings suggest that future policies focusing on the top issues may yield considerable benefit depending on their cost of control.
Although ranking environmental issues is useful in the sense of highlighting priority issues and indicating if there is any surprise environmental issues for the Netherlands. It is important to note that the benefit estimates offer only some guidance on environmental priorities, in the absence of data on costs of implementing policies only part of the picture necessary for establishing priorities is provided. For a full-scale economic analysis benefit estimates need to be compared with cost estimates within a CBA framework. This is outside the scope of this study, however a separate paper on the issues relating to and experience with such CBAs is presented in Annex II.

Preface

This study has been written by a multi-disciplinary team composed of environmental economists from Economics for the Environment (EFTEC) and scientist, economists and modellers at National Institute of Public Health and the Environment (RIVM) for the Ministry of Economic Affairs, the Netherlands, in May 2000. The Ministry of Economic Affairs' aim, to further examine the potential benefit estimates as a guiding tool in environmental policy, was the basis for commissioning this study. The main report is an assessment of the environmental damage due to seven environmental issues in the Netherlands. Damage estimates can be interpreted as benefit estimates of environmental control and can be used as a tool to facilitate an environmental priority scheme for the Netherlands. Annex II presents a paper, written by Professor David Pearce, that examines the role of cost-benefit analysis in efficient decision-making.

The assessment in the main report is new and refreshing for the Netherlands and indeed improves understanding of the potential of benefit estimates as a guiding tool in environmental policy.

Bilthoven, March 2001.

Contents

Abbreviations

Δ	Change in
AOT40	Accumulated ozone above threshold 40ppb, (usually for crops)
AOT60	Accumulated ozone above threshold 60ppb, (usually for health)
BT	Benefits transfer
CBA	Cost benefit analysis
CLS	Current legislation scenario
CO	Carbon monoxide
CO_2	Carbon dioxide
COI	Cost of illness
COPD	Chronic obstructive pulmonary disease
CVM	Contingent valuation methodology
DALY	Disability adjusted life years
dB(A)	Decibel exposure level of noise
D/ERF	Dose / exposure response functions
EU	European Union
GDP	Global damage potential
GDP	Gross domestic product
GNP	Gross national product
GHG	Greenhouse gases
GWP	Global warming potential
IPCC	Intergovernmental Panel on Climate Change
MWTP	Marginal willingness to pay
N	Nitrogen
n	Noise
N_2O	Nitrous oxide
NH_3	Ammonia
NO_2	Nitrogen dioxide
NO_x	Oxides of nitrogen
NEO5	Fifth National Environmental Outlook (*draft report*) (final July 2000)
NSDI	Noise sensitivity depreciation index
O_3	Low level ozone, otherwise known as tropospheric ozone
P	Phosphorous
p.a.	Per annum
PB	Primary benefit
PM10	Fine particles less than 10μm in diameter
PM2.5	Fine particles less than 2.5μm in diameter
POP	Population
pp	Per person
ppb	parts per billion
PPP	Purchasing power parity
RAD	Restricted activity day
RHA	Respiratory hospital admission
SO_2	Sulphur dioxide
UNECE	United Nations Economic Commission for Europe
VOCs	Volatile organic compounds
VOLY	Value of life year
VOR	Value of risk
VOSL	Value of statistical life
WTP	Willingness to pay
Y	Income

Samenvatting

Achtergrond

Het doel van deze studie is het stellen van mogelijke prioriteiten voor het Nederlands milieubeleid. De studie is uitgevoerd door het Economics for the Environment Consultancy (EFTEC) in samenwerking met het Rijksinstituut voor Volksgezondheid en Milieu (RIVM) in opdracht van het Ministerie van Economische Zaken.

Het rapport beschrijft drie verschillende methoden om prioriteiten te stellen binnen het milieubeleid:

- Schadeschatting voor de huidige status van zeven milieuproblemen (1995) en de verwachtte toekomstige ontwikkeling (2010, 2020 en 2030). Schadeschattingen geven een indicatie voor de potentiële baten van milieumaatregelen, met andere woorden de voorkomen schade is gelijk aan de baten van milieumaatregelen;
- Publieke opinie als maatstaf voor het belang van milieuproblemen, zoals waargenomen bij de Nederlandse bevolking, en
- ‘Disability adjusted life years’ (DALY’s).

Milieuproblemen

Het rapport richt zich op zeven milieuproblemen. Deze problemen zijn:

- Klimaatverandering;
- Verzuring;
- Troposferische ozon;
- Fijn stof;
- Geluid;
- Eutrofiëring, en
- Bodemverontreiniging.

Deze onderwerpen zijn tot prioriteit verkozen door de stuurgroep om twee hoofdredenen:

i) Momenteel is het beleid voor deze onderwerpen of niet op zijn plaats of niet geheel effectief[1], en

ii) De verwachting is dat de geselecteerde onderwerpen in Nederland in belang zullen toenemen in de komende decennia.

De data zijn afkomstig uit de concept versie van de Nederlandse Nationale Milieuverkenning 5 (definitieve versie beschikbaar augustus 2000). Er is gekozen voor het ‘EC’ scenario, wat hier wordt aangeduid met ‘current legislation scenario’ (CLS). De toekomstige ontwikkelingen van de milieuproblemen zijn gebaseerd op maatschappelijke trends gecombineerd met het huidige milieubeleid, zoals het reeds vastgesteld is in Nederland en de EU. *Tabel 1* geeft de aannames die ten grondslag liggen aan het CLS.

[1] De stuurgroep heeft besloten om bodemverontreiniging in de studie op te nemen ondanks het huidige beleid dat bodemverontreiniging beperkt. Dit is gedaan om te kijken wat de prioriteit van bodemverontreiniging is in vergelijking met de andere milieuproblemen.

Tabel 1 Maatschappelijke trends en milieubeleid in het CLS

Maatschappelijke trends
• Opkomst van 'Fortress America' en de trend dat strategische handel en industrieel beleid significant bijdragen aan het vormen van handelsblokken; • Ondanks toenemende gespannen relaties met de USA ontwikkelt West-Europa zich erg gunstig. Het Europese proces van integratie is een belangrijke stimulans voor een versterking van de structuur van het West-Europese produkt en arbeidsmarkt. Een verreikend proces van hervorming van de West Europese welvaartsstaat wordt in beweging gezet. Hierin worden pogingen gedaan om de Europese traditie van sociale gelijkheid te combineren met een toegenomen gevoeligheid voor economische stimulansen; • De EU introduceert een energieheffing van $ 10 per barrel; • Technologische ontwikkeling en verspreiding is gematigd; • Hoge migratie naar de EU.
Belangrijk milieubeleid in het CLS*
• Klimaatbeleid (1999); invoering van het Kyoto protocol; • Europese emissies instructies (e.g., EURO IV); • Meest recente normen voor emissie bij verbranding; • Geïntegreerd beleid voor de reductie van ammoniak en mest; • Meest recente geluidsnormen voor transport.

* Beleid goedgekeurd door het Nederlands parlement voor 1 januari 2000

Methoden

Schadebenadering

De toegepaste methode komt grotendeels overeen met de methode die gevolgd is voor de studie 'European Environmental Priorities: an integrated economic and environmental assessment' (RIVM et al., 2000)[2] voor de Europese Commissie DG Milieu. De methode is gebaseerd op een logische stapsgewijze opeenvolging van emissies, verandering in blootstelling, kwantificeren van effecten met behulp van blootstellings-effect relaties, tot waardering gebaseerd op 'willingness-to-pay' (WTP).

We onderkennen het bestaan van significante onzekerheid bij het schatten van milieuschade als gevolg van:

- Statistische fout;
- Overbrengen van blootstellings-effect relaties en waarderingen naar een andere context (locatie en tijd) ;
- Variatie in politieke en ethische opvattingen, en
- Tekortkomingen in het huidige kennisniveau, in sommige gevallen leidend tot het weglaten van effecten.

We benaderen het bestaan van onzekerheid door het zoveel mogelijk kwantificeren van effecten, gebruikmakend van wat wij de beste beschikbare data vinden (na een uitgebreide bestudering van de literatuur), en de aannames die zoveel mogelijk overeenkomen met deze data. Wij anticiperen op het bestaan van onzekerheid door het uitvoeren van een gevoeligheidsanalyse om op een overzichtelijke manier de gevolgen van de onzekerheid op de robuustheid van onze conclusies, gebaseerd op onze baseline data en aannames, weer te geven. Om een duidelijk overzicht te bewaren is er een gevoeligheidsanalyse uitgevoerd voor elk milieuprobleem.

De belangrijkste bronnen van onzekerheid, zoals vastgesteld in loop van deze studie, zijn de volgende:

[2] Met uitzondering van bodemverontreiniging, welke niet was opgenomen in deze studie. Voor een uitgebreide uiteenzetting van de ontwikkelde en gebruikte methode voor dit onderwerp, zie *Section 4.4*.

- Benadering van de waardering van vroegtijdige sterfte;
- De 'willingness-to-pay' waarden worden constant verondersteld, in Euro 2000 waarden, over de gehele tijdsperiode ondanks een stijgend Nederlands BBP.
- De relatie tussen blootstelling en uiteindelijke gezondheidseffecten, oftewel de blootstellings-effectrelaties;
- Het beleid ter voorkoming van klimaatverandering kan hogere baten voortbrengen wanneer de secundaire baten als gevolg van andere milieuproblemen worden meegenomen. Aan de andere kant kan de schade overschat worden door het weglaten van aanpassingsstrategieën;
- De baten van verzuring kunnen onderschat zijn door het weglaten van de effecten op ecosystemen, cultuurgoederen en zichtbaarheid. De baten van verzuring kunnen overschat worden, doordat de effecten van PM_{10} op de volksgezondheid worden meegenomen, terwijl deze effecten al in de separate analyse voor PM_{10} berekend worden;
- De aanname voor geluidhinder is dat alle type geluid hetzelfde gewaardeerd worden, ondanks het bewijs dat veronderstelt dat geluid van vliegtuigen en railverkeer als 'erger' beschouwd wordt dan geluid als gevolg van wegverkeer;
- De baten van bodemverontreiniging zijn behoorlijk onzeker als gevolg van de data met betrekking tot het aantal verontreinigde locaties, het omzetten van aantal locaties in omvang verontreinigde grond, en de waarde van schone / verontreinigde grond;
- De baten van fijn stof worden geschat op basis van de aanname dat alle fracties van PM_{10} even schadelijk zijn voor de volksgezondheid en het feit dat in de resultaten andere ziekte-effecten dan ziekenhuisopnames niet zijn opgenomen;
- Eutrofiëring; aanzienlijke onzekerheid omtrent de wetenschappelijke data voor de waterkwaliteit in Nederland en het gebrek aan bewijs voor een WTP voor daling van de eutrofieringseffecten voor binnenwateren in Nederland, en
- Troposferische ozon; niet meegenomen zijn de effecten op materialen, bosecosystemen, niet-gewas begroeiing en biodiversiteit, en de ziekte effecten anders dan ziekenhuisopnames.

Deze en andere bronnen zijn vollediger uiteengezet en onderzocht in het rapport.

Sommige criticie beweren dat het bestaan van onzekerheid de betrouwbaarheid van een batenschatting of de batenschatting als een beslissingsinstrument ondermijnt. Het is onze professionele opvatting dat de aanwezigheid van een grote onzekerheid het essentiëler maakt om een batenschatting uit te voeren. Een batenschatting vergroot de kennis in het probleemgebied en het geeft politicie een indicatie voor het potentiële risico van hun acties. Een alternatieve benadering is dat alleen de baten waarvan de begeleidende onzekerheid als minimaal gekwantificeerd is, worden meegenomen. Echter dit zou betekennen dat het noodzakelijk is om een subjectief standpunt in te nemen met betrekking tot hoe goed het bewijs moet zijn om een gegeven effect als robuust te beschouwen voor de analyse. Behalve het vaststellen of een vervuiler schadelijk is, geeft het een gebrekkig advies voor de reeks van mogelijke effecten van de onderzochte vervuilers.

Publieke opiniebenadering

Om de belangrijkheid van de milieuproblemen zoals bezien door de Nederlands bevolking te bepalen, refereren we naar Europese en nationale onderzoeken. De redenen om naar publieke opinie te kijken zijn tweeledig;

- Verscheidene Europese en nationale onderzoeken tonen dat het milieu een belangrijke bron van zorg blijft voor de Nederlandse bevolking, en
- Het gebruik van publieke opinie voor het rangschikken van milieuproblemen verzekert dat alle inwoners van Nederland een even hoge weging hebben. Met andere woorden, ze krijgen in feite een even groot aantal 'stemmen' over het milieu. Een dergelijke rangschikking van milieuproblemen is daarom ongevoelig voor verschil in factoren, die een batenschatting kunnen beïnvloeden, zoals bijvoorbeeld inkomen.

Disability adjusted life years (DALY's) benadering

De gebruikte methode komt grotendeels overeen met de methode van Murray en Lopez (1996). Zij ontwikkelden de 'disability adjusted life years' maatstaf om de wereldwijde ziektelast en de daaruitvolgende gezondheidsbeleidsprioriteiten in verschillende regio's van de wereld te schatten. Deze gezondheidseffectmaatstaf combineert verlies van levensjaren en jaren geleefd met ziekte of handicap, die gewogen zijn naar zwaarte.

In het kader van de 5e Nationale Milieuverkenning zijn alleen adequate data en toekomstperspectieven beschikbaar voor fijn stof, troposferische ozon, geluid, ultra-violette straling, radon, huisvochtigheid en ziekte als gevolg van voedselinfecties. Voor elke relevante gezondheidsuitkomst berekenen we toegeschreven risico's door het combineren van populatie gewogen blootstellingverdeling met relatieve risicoschattingen, afgeleid uit de epidemiologische literatuur. Vervolgens is voor iedere gezondheidsuitkomst het aantal gevallen geschat door het combineren van de in de baseline voorkomende gevallen met de toegevoegde risico's. Berekeningen van de toekomstige ziektelast zijn gebaseerd op projecties van de toekomstige populatiestructuur. Wij presenteren de gebruikte set van eindpunten om te komen tot de schattingen van toegeschreven ziektelast en het aantal verloren DALY's. Tenslotte is de totale blootstelling toegeschreven aan ziektelast berekend door het aggregeren van het aantal DALY's voor elke gezondheidsuitkomst. De ziektelast veroorzaakt door additionele UV-blootstelling, als gevolg van degradatie van de ozonlaag, is berekend door het aggregeren van jaarlijkse ziekte en sterfte schattingen van huidkanker en de Nederlandse ziektelast data, Melse et al (2000). Statistische onzekerheid is geschat met MonteCarlo technieken.

Resultaten

Schadebenadering

Om voor de milieuproblemen op basis van schade (of potentiële baten van beleid) prioriteiten te stellen, moeten de schatting direct vergelijkbaar gemaakt worden. Dit wordt gecompliceerd door het feit dat de schadeschattingen voor geluid en bodemverontreiniging contante waarden over een oneindige periode zijn. Dus we vergelijken de milieuproblemen met de contante waarde van de schadeschatting (rente = 6%). In *Tabel 2* worden de totale schadeschattingen gegeven als een netto contante waarde en de overeenkomende waarde voor de jaarlijkse schade, vervolgens zijn de Nederlandse milieuthema's gerangschikt naar de hoogste potentiële baten van beleid.

Tabel 2 Totale en jaarlijkse schadeschattingen voor milieuthema's in Nederland

	Totale schade Netto contante waarde rentevoet=6% Miljoen Euro(2000)	Jaarlijkse schade miljoen Euro (2000)	Rangschikking
Troposferische ozon	110034 – 110613	6228 - 6261	1
Bodemverontreiniging	59559	3371	2
PM_{10}	54471	3083	3
Verzuring	37017 - 41569	2095 - 2353	4
Klimaatverandering	36766	2081	5
Geluid	31980	1810	6
Eutrofiëring	9835 - 19224	557 - 1088	7

Dus we zien dat de hoogste prioriteit, in termen van potentiële baten van volledig beleid, ligt bij troposferische ozon, bodemverontreiniging en fijn stof. Gevolgd door klimaatverandering en verzuring, terwijl voor geluid en eutrofiëring geschat wordt dat ze de laagste potentiële baten zullen opleveren. Maar er kunnen geen conclusies ten aanzien van de wenselijkheid van beleid getrokken worden, omdat kostenschattingen ontbreken.

Ondanks dat de schade en dus de potentiële primaire baten van beleid voor milieuproblemen toenemen in de tijd (met uitzondering van eutrofiëring en verzuring), dalen ze als percentage van het Nederlands BBP in 1995, 2010, 2020 and 2030. *Tabel 3* geeft de schadeschattingen als percentage van het BBP. Aangemerkt moet worden dat dalende percentages van het BBP geen garantie geven voor een stijging van de relatieve waarde van het milieu in de tijd als het inkomen stijgt. Als de relatieve waarde met hetzelfde percentage stijgt als het BBP, dan blijft de schade als deel van het BBP gelijk. Maar er is slechts weinig informatie beschikbaar over de inkomenselasticiteit van de vraag naar milieu, dus we nemen aan dat 'willingness to pay' (WTP) waarden constant blijven in de tijd ondanks een stijgend BBP voor Nederland.

Tabel 3 Milieuschadeschattingen als percentage van Nederlands BBP: %

	1995	*2010*	*2020*	*2030*
Troposferische ozon	1,32	1,12	1,01	1,02
PM_{10}	0,76	0,53	0,47	0,46
Klimaatverandering	0,62	0,40	0,33	0,27
Verzuring	0,85 – 0,95	0,30 – 0,34	0,23 – 0,25	0,18 – 0,20
Eutrofiëring	0,22 – 0,45	0,08 – 0,16	0,06 – 0,12	0,05 – 0,09
Geluid	0,51	0,38	0,30	0,25
Totaal	4,29 – 4,62	2,82 – 2,92	2,40 – 2,48	2,22 – 2,29

Nederlands BBP (miljard Euro): 1995 = 312,627; 2010 = 482,543; 2020 = 632,928; 2030 =830,180; bron RIVM (2000).

Tabel 3 suggereert dat milieuschade in Nederland een significant aandeel is van het BBP in 1995, een spreiding van 1.3% voor tropsferische ozon, tussen 0.6% en 0.8% voor fijn stof, klimaatverandering, verzuring tot ruwweg 0.5% voor geluid en eutrofiëring. In totaal is de milieuschade als gevolg van bovenstaande milieuproblemen geschat op ruwweg 4.5% van het BBP in 1995, dalend tot ongeveer 2% in 2030. Het is interessant om deze getallen te vergelijken met de schattingen voor uitgaven aan vervuilingbestrijding, waarvan aangegeven is dat ze ongeveer 1.2% van het BBP in 1990 zijn (ERECO, 1992).

Publieke opiniebenadering

Ondanks dat het milieu minder als een probleem wordt gezien in 1999 dan in 1995, wanneer de bezorgdheid op haar hoogtepunt was, (63% in 1986, 80% in 1995 en 70% in 1999), (Eurobarometer 1986, 1995 en 1999), blijft het een punt van gemeenschappelijke zorg voor het Nederlandse publiek. *Tabel 4* laat de onderwerpen zien die serieuze bedreiging voor het milieu vormen (ongeacht de schaal), zoals waargenomen bij het Nederlands publiek.

Deze opinies zijn tamelijk stabiel in de tijd en de resultaten van 1992 geven eenzelfde beeld (behalve het onderwerp zure regen, dat iets belangrijker geworden is sinds 1992).

Tabel 4 Onderwerpen die bijdragen aan serieuze milieuschade: 1992 and 1995

Onderwerp	*Rangschikking*
Fabrieken die gevaarlijke chemicaliën uitstoten in lucht en water	1
Olievervuiling van zeeën en kusten	2
Wereldwijde vervuiling (geleidelijke verdwijning van tropisch regenwoud, afbraak van de ozonlaag, broeikaseffect etc)	3
Opslag van radioactief afval	4
Industrieel afval	5
Zure regen	6
Overmatig gebruik van herbiciden, insecticiden en kunstmest in de landbouw	7
Zwerfvuil op straat, in groene gebieden of op het strand	8
Luchtverontreiniging door auto's	9
Ongecontroleerd massa toerisme	10
Afvalwater	11
Geluid ontstaan door openbare gebouwen of werken, zwaar verkeer, luchthavens	12

Bron: Eurobarometer: Europeans and their Environment, 1992, 1995 en SCP onderzoek 1993.

Disability adjusted life years (DALY's) benadering
De gezondheidseffecten van luchtverontreiniging door fijn stof en gedeeltelijk door tropesferische ozon, domineren de ziektelast. De toekomstige ziektenlast is voor een groot deel het gevolg van toekomstige veranderingen in de populatiestructuur (een grotere groep oudere mensen wordt beïnvloed door dit type luchtverontreiniging). Een ander milieuprobleem dat geassocieerd wordt met een hoge ziektelast is blootstelling aan geluid van weg- en luchtverkeer. We laten na om ziektenlast toe te schrijven aan het hoge aantal mensen dat serieuze hinder en slaapproblemen aangeeft.. Er is namelijk veel discussie of deze respons gezien moet worden als schade aan de volksgezondheid of meer als een sociale reactie. In plaats daarvan schatten we de mogelijke fractie van cardiologische ziekten die toe te wijzen zijn aan blootstelling aan geluid op basis van de resultaten van diverse omvangrijke epidemiologische studies die een causaal verband impliceren. De ziektelast als gevolg van de overgebleven milieuproblemen zijn verhoudingsgewijs miniem. *Tabel 5* geeft het jaarlijks verlies aan DALY's als gevolg van de geselecteerde milieuproblemen.

Tabel 5 Jaarlijks verlies aan Disability-Adjusted Life Years (DALY's) als gevolg van de geselecteerde milieuthema's

Onderwerp	DALY's in 2030 (mediaan)	Onzekerheidsmarge	Rangschikking
Fijn stof	3300	1600 - 5400	1
Aangeboren voedsel infectie ziekten	2800	1850 - 4000	2
Geluid*	2600	700 - 4400	3
Troposferische ozon	1900	600 - 4100	4
Radon binnenshuis	1700	500 - 4100	5
Vochtigheid binnenshuis	400	200 - 700	6
UV straling	350	-	7

* weg- en luchtverkeer; vergelijkt alleen klinische gezondheidsuitkomsten

Conclusies

Het rangschikken van milieuproblemen is nuttig om onderwerpen die prioriteit hebben te onderstrepen en om aan te geven of er een verrassende uitkomst is voor de Nederlandse milieuproblemen. Zulke studies kunnen gebruikt worden om het bewustzijn van de verantwoordelijke mensen te vergroten. Ondanks dat de rangschikking geen enkele politieke vraag beantwoordt, (hiervoor moeten naast de baten ook de kosten van de maatregelen berekent worden) kunnen de berekende eenheidswaarden van de studie gebruikt worden voor een toekomstige kosten-batenanalyse voor milieumaatregelen.

Het is belangrijk om op te merken dat de schade of batenschattingen, zoals gepresenteerd voor de verschillende milieuproblemen slechts een gedeeltelijke indicatie geven voor de milieuprioriteiten in Nederland. Door het ontbreken van data voor de kosten van de implementatie van het beleid, kunnen deze maatstaven van effictiviteit slechts een deel van het plaatje, wat noodzakelijk is voor het vaststellen van prioriteiten, geven. Voor een volledige economische analyse zoals in RIVM et al (2000), moeten baten(schade)schattingen aan kostenschattingen gekoppeld worden in een kosten-baten analyse schema. Dit valt buiten het bereik van deze studie, maar een apart rapport over de onderwerpen in relatie tot en ervaring met kosten-baten analyse wordt gepresenteert als Annex II van de totale studie (zie *Annex II: Integrating Cost Benefit Analysis into the Policy Process*).

Het is ook belangrijk om het verschil aan te geven tussen de schade van een milieuthema, zoals klimaatverandering en de baten van beleidsmaatregelen om klimaatverandering te voorkomen. De baten bevatten de voorkomen schade maar zijn waarschijnlijk significant hoger door bijkomende voordelen van milieubeleid. Deze bijkomende voordelen zijn beter bekend als secundaire baten. Secundaire baten ontstaan omdat maatregelen voor een bepaald milieuprobleem ook andere vervuilende stoffen reduceren. Klimaatbeleid zal behalve het verminderen van broeikasgassen ook de uitstoot van verzurende stoffen verminderen. Dus de baten van beleidsmaatregelen voor een milieuprobleem zullen waarschijnlijk de gemaakte schatting overstijgen. Een belangrijk onderwerp is wanneer de primaire en secundaire baten optreden, nu of in de toekomst, en het effect van disconteren. De secundaire baten van bijvoorbeeld klimaatbeleid zullen dichterbij het heden plaatsvinden dan de primaire baten die ver in de toekomst zullen plaatsvinden. Een ander belangrijk onderwerp is dat verwacht wordt dat de secundare baten van klimaatbeleid (i.e. SO_x, NO_x) zullen dalen in de toekomst, omdat ze afhankelijk zijn van klimaatsonafhankelijk beleid dat leidt tot daling van emissies.

Om een definitieve rangschikking naar de belangrijkheid van de Nederlands milieuproblemen te maken, gebruiken we de resultaten van de schadeberekeningen, de publikie opinie in Nederland en de DALY-benadering. *Tabel 6* brengt de resultaten van de drie methoden samen.

De definitieve rangschikking van milieuproblemen, zoals gegeven in de één na laatste kolom van *Tabel 6* is gebaseerd op een eenvoudige 'Borda count'. Dit betekent dat voor ieder milieuprobleem we de gewogen rangschikking van de schadeberekening, de publieke opinie en de DALY-benandering optellen en delen door het totaal aantal beschouwde milieuproblemen. Het is noodzakelijk om de rangen te wegen om zodoende de verschillende aantallen beschouwde milieuproblemen in de verschillende benaderingen mee te nemen. De algehele rangschikking wordt gevonden met behulp van de resultaten van de 'Borda count', waar een lage waarde een hoge prioriteit scoort. Ondanks dat de 'Borda count' een conventionele manier is om een aantal rangschikkingen te rangschikken is het grootste nadeel van deze procedure dat de algehele rangschikking niet gevoelig is voor de onzekerheid van de verschillende milieuproblemen. Om de definitieve rangschikking te kwalificeren bevat de laatste kolom van *Tabel 6* een benadering van de algehele onzekerheid van elk milieuprobleem met een schaal van ++ tot --, waar ++ een lage onzekerheid aangeeft en -- een hoge onzekerheid.

Tabel 6 Milieuthema's in Nederland in volgorde van prioriteit

Milieuprobleem	Rangschikking volgens schade-berekening	Rangschikking volgens publieke opinie	Rangschikking volgens DALY-benadering	Definitieve rangschikking	Onzekerheid
Bodemverontreiniging	2	-	-	1	--
Klimaatverandering	5	3	-	2	++
PM_{10}	3	9	1	3	+
Verzuring	4	6	-	4	++
Troposferische ozon	1	9	4	5	+
Eutrofiëring	7	7	-	6	--
Geluid	6	12	3	7	-
Aantal beschouwde onderwerpen in de studie	*7*	*12*	*7*	-	

Deze studie concludeert dat in volgorde van prioriteit, bodemverontreiniging, klimaatverandering en fijn stof de top drie prioriteit van milieuproblemen in Nederlands zijn, gevolgd door verzuring, troposferische ozon, eutrofiëring en tenslotte geluid. Deze bevindingen suggereren dat toekomstig beleid gericht op de onderwerpen met een top prioriteit aanzienlijke baten kunnen opbrengen.

Summary

Background
The objective of this study is to set priorities for the environmental policy in the Netherlands. The study is undertaken by Economics for the Environment Consultancy (EFTEC) with Rijksinstituut voor Volksgezondheid en Milieu (RIVM) for the Ministry of Economic Affairs, the Netherlands.

This report describes three different approaches to environmental policy prioritisation:
- Damage assessment for the current status of seven environmental issues (1995) and the expected future progress (2010, 2020 and 2030). Damage estimates indicate what the benefits of environmental control could be, i.e. avoided damage equals benefit of control;
- Public opinion, as a measure of the importance of environmental issues as perceived by the population of the Netherlands, and
- 'Disability adjusted life years' (DALYs).

Environmental issues
The report focuses on seven environmental issues. These are:
- Climate change;
- Acidification;
- Low level ozone;
- Particulate matter;
- Noise;
- Eutrophication, and
- Land contamination.

These issues are chosen as priorities by the steering group for two main reasons:
i) At present the regulatory systems for these issues are either not in place or not wholly effective[3], and
ii) The issues listed are expected to be increasing in importance in the next decades in the Netherlands.

The data are drawn from the Fifth National Environmental Outlook for the Netherlands (NEO5) draft report. The 'medium growth' scenario is chosen and this is referred to here as the 'current legislation scenario' (CLS). The future growth of environmental issues is based on societal trends combined with current environmental policies already in place in the Netherlands and the EU. *Table 1* presents the assumptions behind the CLS.

[3] Although policies are in place to control land contamination, the steering group decided to include land contamination in the priority assessment in order to see how land contamination compares with the other environmental issues in the Netherlands, in terms of priority.

Table 1	*Societal trends and environmental policies included in the CLS*
Societal trends	
• Rise of 'Fortress America' and the tendency toward strategic trade and industrial policies significantly contribute to formation of trade blocks; • Despite increasingly strained relations with the USA, Western Europe develops very favourably. The European process of integration is an important stimulus toward strengthening incentive structures in the Western European product and labour markets. A far-reaching process of reform of the Western European welfare state is set in motion. In this, attempts are made to combine the European tradition of social equity with an increased sensitivity to economic incentives; • EU introduces an energy/carbon tax of $ 10 per barrel; • Technological development and diffusion is moderate; • High migration to the EU from outside the EU.	
Key environmental policies included in the CLS*	
• Climate change policy plan (1999); implementation of Kyoto protocol; • European emission directives (e.g., EURO IV); • Most recent emission standards for combustion; • Integrated policy plan for reducing ammonia and manure; • Most recent noise standards for transport.	

* Policies in place as approved by the Dutch parliament before January 1, 2000

Methodology

Damage assessment approach

The methodology adopted largely follows that of the European Commission DG Environment study 'Economic Assessment of Priorities for a European Environmental Policy Plan' (RIVM et al., 2000)[4]. It is based on a logical stepwise progression through emission, change in exposure, quantification of impacts using exposure-response functions, to valuation based on willingness-to-pay.

We acknowledge the existence of significant uncertainty in assessment of environmental damage arising through:

- Statistical error;
- Transfer of exposure-response functions and valuations from one context (location and time) to another;
- Variation in political and ethical opinion, and
- Gaps in current knowledge base, leading in some cases to omission of effects.

Our approach to the existence of uncertainty is to quantify effects as far as possible using what we regard (from a comprehensive review of the literature) to be the best data available, and assumptions which correspond most closely with those data. We respond to the existence of uncertainty through a sensitivity analysis to demonstrate in a transparent manner the consequences of uncertainty on the robustness of our conclusions based on our baseline data and assumptions. To retain transparency a sensitivity analysis is conducted for each environmental problem.

The following is a detailed account of sources of uncertainty:

- Approach to the monetary valuation of premature mortality;
- Willingness-to-pay values assumed to remain constant, at Euro 2000 values, through time despite increasing GDP for the Netherlands;
- Relationships between exposure and health end points, i.e. dose / exposure response functions;

[4] With the exception of land contamination, which was not included in that study. For a detailed discussion of the methodology developed and implemented for this issue see *Section 4.4*.

- Climate change control policies may yield greater benefits if the secondary benefits to other environmental issues are included. Damage estimates may however be overstated due to the omission of adaptation strategies;
- Acidification benefits may be understated due to the omission of effects on ecosystems, cultural assets and impacts to visibility. Acidification benefits may be overestimated due to the inclusion of impacts due to PM_{10} on human health which are already accounted for in the separate analysis on PM_{10};
- Noise nuisance assumption, that all noise types are valued the same despite the evidence that suggests aircraft and rail noise may be more 'annoying' than road noise.
- Land contamination benefits are very uncertain due to the uncertain data for number of contaminated sites, the conversion of 'number of contaminated sites' to size of contaminated land, and the value of clean and contaminated land;
- Particulate matter benefit estimates are based on the assumption that all fractions of PM_{10} are equally aggressive to human health and the results omit morbidity effects other than hospital admissions;
- Considerable uncertainty is attached to the scientific data for water quality in the Netherlands used for eutrophication damage estimates. There is also a lack of evidence of a WTP for a reduction of eutrophication impacts for inland waters in the Netherlands, and
- The omission of impacts due to low level ozone to materials, forests ecosystems, non-crop vegetation and biodiversity and the morbidity effects other than hospital admissions is likely to lead to underestimation.

These and other sources of uncertainty are more fully discussed and investigated in the report.

Some commentators argue that the existence of uncertainty undermines the credibility of the benefit estimates as a decision making tool. It is our professional opinion that the presence of large uncertainty makes it more essential that benefit assessment is conducted. It serves to increase the knowledge base in the area of question and it acts as a signal to policy makers for the potential risks of their actions. An alternative to the approach adopted here would be to quantify only those benefits for which associated uncertainty is minimal. However, this would mean taking a necessarily subjective position on how good the evidence must be on a given effect for analysis to be considered robust. Beyond establishing if a pollutant is known to be harmful, it would provide poor guidance on the range of possible effects of the pollutants considered.

Public opinion approach

In order to determine the importance of environmental issues as perceived by the population of the Netherlands we refer to European and national surveys. The rationale for turning to public opinion is twofold;

- Various European and national public opinion surveys show that the environment remains a major concern for the Dutch public, and
- Using public opinion to rank environmental issues ensures all Dutch citizens are weighted equally. In other words, they are effectively given an equal number of ‘votes’ on the environment. Such a ranking of environmental issues is therefore impartial to differences in factors, such as income, that can affect economic assessments.

Disability adjusted life years (DALYs) approach

The DALY methodology largely follows that of Murray and Lopez (1996). They develop the ‘disability adjusted life years’ measure in order to assess the global disease burden and consequently the health policy priorities in different regions of the world. This health impact measure combines years of life lost and years lived with disease or disability that are weighted according to severity.

In the NEO5 framework adequate data and future projections are available for particulate matter, low level ozone, noise, ultra-violet radiation, radon, home dampness and food borne infectious disease only. For each relevant health outcome we calculate attributable risks by combining population weighted exposure distributions with relative risk estimates derived from the epidemiological literature. Subsequently for each health outcome the number of cases was estimated by combining baseline incidence rates with the attributive risks. Calculations of future disease burden are based on projections of future population structure. We present the set of endpoints used to arrive at estimates of attributable disease burden and the number of DALYs lost. Finally a total exposure attributable disease burden was calculated by aggregating the number of DALYs for each health outcome. The disease burden associated with additional UV-exposure due to ozone layer degradation was calculated by aggregating annual morbidity and mortality estimates of skin cancer and Dutch burden of disease data, (Melse et al., 2000). Statistical uncertainty was assessed using Monte Carlo techniques.

Results

Damage assessment approach

In order to prioritise the environmental issues in order of damages (or potential benefits from control), the damage estimates must be made directly comparable. This is complicated by the fact that damage estimates for land contamination are present values. Thus, we compare the environmental issues according to present value damage estimates (discount rate = 6%). *Table 2* gives the total damage estimates as a present value and the corresponding annual damage value and then the environmental issues for the Netherlands are ranked in terms of greatest potential benefit from control.

Table 2 Total and annual damage estimates for environmental issues in the Netherlands

	Total damage Present value, discount rate=6% Euro million (2000)	Annual damage Euro million (2000)	Ranking
Low level ozone	110034 - 110613	6228 – 6261	1
Land contamination	59559	3371	2
PM_{10}	54471	3083	3
Acidification	37017 - 41569	2095 – 2353	4
Climate change	36766	2081	5
Noise	31980	1810	6
Eutrophication	9835 - 19224	557 – 1088	7

Thus we see that the priorities, in terms of potential benefits from full control, are low level ozone, land contamination and particulate matter, followed by acidification and climate change, whilst noise and eutrophication are estimated to yield the lowest potential benefits from control. However, in the absence of cost estimates no conclusions can be reached on the desirability of control measures. The full discussion on this methodology and the results is given in *Section 3 and 4*.

Despite the fact that damages and hence the potential primary benefits of control are rising over time for the environmental issues (with the exception of eutrophication and acidification), they fall as a proportion of Dutch GDP in 1995, 2010, 2020 and 2030. *Table 3* presents the damage estimates as a percent of GDP. Note however that the falling percentage of GDP results makes no allowance for a rising relative value of the environment over time as income rises. If these relative valuations rise at the same rates as GDP, the proportion of damage to GDP would remain the same. Little information is available on the income elasticity of demand for the environment, thus we assume that ‘willingness to pay’ (WTP) values are constant through time despite increasing GDP for the Netherlands.

Table 3 Environmental damage estimates as a percent of Dutch GDP: %

	1995	*2010*	*2020*	*2030*
Low level ozone	1.32	1.12	1.01	1.02
PM10	0.76	0.53	0.47	0.46
Climate change	0.62	0.40	0.33	0.27
Acidification	0.85 - 0.95	0.30 - 0.34	0.23 - 0.25	0.18 - 0.20
Eutrophication	0.22 – 0.45	0.08 – 0.16	0.06 – 0.12	0.05 – 0.09
Noise	0.51	0.38	0.30	0.25
Total	4.29 - 4.62	2.82 - 2.93	2.40 - 2.48	2.22 - 2.29

Dutch GDP: Euro billion: 1995 = 312.627, 2010 = 482.543, 2020 = 632.928 and 2030 = 830.180, source RIVM (2000).

Table 3 suggests that environmental damage in the Netherlands is a significant proportion of GDP in 1995, ranging from 1.3% for low level ozone, between 0.6% and 0.8% for PM_{10}, climate change, acidification and roughly 0.5% for noise and eutrophication. Overall, total environmental damage due to the above environmental issues is estimated to be roughly 4.5% GDP in 1995, falling to about 2% in 2030. It is interesting to compare these figures with the estimates of expenditure on pollution abatement, reported to be about 1.2% of GDP in 1990 (ERECO, 1992).

Public opinion approach

Although the environment was seen as less of a problem in 1999 than in 1995, when concern was at its highest, (63% in 1986, 80% in 1995 and 70% in 1999), (Eurobarometer 1986, 1995 and 1999), it remains a common concern for the Dutch public. *Table 4* presents the issues considered to constitute a ‘serious threat to the environment’ (regardless of locality) as perceived by the Dutch public.

Table 4 Issues considered by Dutch public to constitute serious environmental damage: 1992 and 1995

Issue	*Ranking*
Factories releasing dangerous chemicals into the air or water	1
Oil pollution of the seas and coasts	2
Global pollution (gradual disappearance of tropical forests, destruction of the ozone layer, greenhouse effect etc)	3
Storage of nuclear waste	4
Industrial Waste	5
Acid Rain	6
Excessive use of herbicides, insecticides and fertilisers in agriculture	7
Rubbish in the streets, in green spaces or on beaches	8
Air pollution from cars	9
Uncontrolled mass tourism	10
Sewage	11
Noise generated by building or public works, heavy traffic, airports	12

Source: Eurobarometer: Europeans and their Environment, 1992, 1995 and SCP survey 1993.

These opinions are fairly resilient to time and results from 1992 show similar rankings (excepting the issue of acid rain, which has increased in importance slightly since 1992). The full discussion on these results is presented in *Section 5*.

Disability adjusted life years (DALYs) approach

The health effects of air pollution from particulate matter, and to a certain degree from low level ozone, dominate the disease burden. The future disease burden is to a large extent the result of future changes in the population structure, i.e. a greater share of older people are affected by this type of air pollution. Another environmental problem associated with a high disease burden is noise exposure from road and air traffic. We refrained from attributing the disease burden to the large number of people reporting serious annoyance and sleep disturbance. There is much discussion about whether these responses should be regarded as a damage to human health or merely a social response. Instead we estimate the possible fraction of cardiovascular disease attributable to noise exposure based on the results of several large epidemiological studies implicating a causal association. The disease burden due to the remaining environmental issues are by comparison relatively minor, a fuller discussion of these issues is presented in *Section 4.8.2*. *Table 5* presents the DALYs lost yearly due to the selected environmental issues.

Table 5 Disability-Adjusted Life Years (DALYs) lost annualy to selected environmental issues

Issue	*DALYs in 2030 (mean)*	*Uncertainty range*	*Ranking*
Particulate air pollution	3300	1600-5400	1
Food borne infectious disease	2800	1850-4000	2
Noise*	2600	700-4400	3
Tropospheric ozone	1900	600-4100	4
Indoor radon	1700	500-4100	5
Home dampness	400	200-700	6
UV radiation	350	-	7

* Road and air traffic; comprises only clinical health outcomes

Conclusions

Ranking environmental issues is useful in the sense of highlighting priority issues and indicating if there is any surprise environmental issues for the Netherlands. Such exercises can be used for awareness raising for decision makers. Although ranking does not answer any questions about policy, in order to do so we would need to compare the benefits of environmental control with the costs, the unit damage values used in the benefit assessment study can be re-used if a CBA of environmental policy is conducted in future.

It is important to note that the damage or benefit estimates presented for the various environmental issues offer only some guidance on environmental priorities for the Netherlands. In the absence of data on costs of implementing policies, these measures of effectiveness can provide only part of the picture necessary for establishing priorities. For a full scale economic analysis, like that in RIVM et al (2000), benefit (damage) estimates need to be compared with cost estimates within a CBA framework. This is outside the scope of this study. However, a separate paper on the issues relating to and experience with such CBAs is prepared as Annex II of the overall study (see Annex *II: Integrating Cost-Benefit Analysis into the Policy Process*).

It is also important to distinguish between damages of an environmental issue, such as climate change and the benefits of policy measures to control the issue. The benefits include the avoided damages but are likely to be significantly greater because of the ancillary gains from environmental policy. These are known as the secondary benefits. Secondary benefits arise because the control of an environmental issue is likely to involve policies, which will also reduce other pollutants, e.g. climate change control policies will reduce greenhouse gases as well as the acidifying pollutants. Thus, benefits of a policy measure to control an environmental issue are, most likely to exceed the estimates of avoided damage. An important consideration is the issue of when primary and secondary benefits take place, i.e. now or sometime far in the future, and the effect of discounting. For example, the secondary benefits of climate change control measures will take place closer to the present, rather than decades or centuries into the future as with the primary benefits. Another important consideration is that since most secondary pollutants of greenhouse gas control policies i.e. SO_x, NO_x, are subject to independent policies, emissions are expected to fall over time, this means that climate change policies will secure further but smaller secondary benefits in the future.

In order to determine a final ranking for the environmental issues in the Netherlands in order of importance we draw upon the results of the damage assessment, public opinion in the Netherlands and the DALY assessment. *Table 6* brings together the results of the three methods.

The final ranking for the environmental issues given in the fifth column of *Table 6* is based on a simple ‘Borda count’, i.e. for each environmental problem we sum the weighted ranking from the public opinion, the damage assessment and the DALY procedure and divide this by the number of total environmental issues considered. It is necessary to weight the rankings in order to allow for the different numbers of environmental issues considered in the different approaches. The overall ranking is found by ordering the results of the ‘Borda count’, where lower values score higher priorities. Although the ‘Borda count’ is a conventional way to rank a number of rankings, the main disadvantage of this procedure is that the overall rankings are not sensitive to the uncertainty associated with each environmental problem. To qualify the final rankings, *Table 6* includes an assessment of overall uncertainty for each problem in the final column, on a scale of ++ to --, where ++ indicates low uncertainty and -- indicates high uncertainty.

Table 6 Environmental issues in the Netherlands in order of priority

Environmental problem	ranking according to damage assessment	ranking according to public opinion	ranking according to DALYs	final ranking	Uncertainty
Land contamination	2	-	-	1	--
Climate change	5	3	-	2	++
PM10	3	9	1	3	+
Acidification	4	6	-	4	++
Low level ozone	1	9	4	5	+
Eutrophication	7	7	-	6	--
Noise	6	12	3	7	-
No of issues considered in each methodology	*7*	*12*	*7*		

This study concludes that land contamination, climate change and particulate matter are the top priority environmental issues in the Netherlands, followed by acidification, low level ozone, eutrophication and finally noise. These findings suggest that future policies focusing on the top issues may yield considerable benefit depending on their cost of control.

1. Background to and scope of the study

In 1998, the Dutch Ministry of Economic Affairs commissioned a research project on the valuation of the benefits of environmental policy. The research steering group concluded that monetary valuation should have a role in environmental policy decision making.

A major outcome of this process is the Dutch Ministry of Economic Affairs' aim to further examine the potential benefit estimates as a guiding tool in environmental policy. The Ministry has a particular interest in:

- Benefit estimates as a policy tool for prioritising environmental policy, and
- Benefit estimates as part of the use of cost-benefit ratios in environmental policy.

As the next step to further examine the potential benefit estimates for various environmental issues in the Netherlands this study was undertaken by Economics for the Environment Consultancy Ltd (EFTEC) with Rijksinstituut voor Volksgezondheid en Milieu (RIVM).

This report focuses on seven environmental issues as chosen by the steering group. These are:

1. Climate change;
2. Acidification;
3. Noise;
4. Land contamination;
5. Particulate matter;
6. Eutrophication, and
7. Low level ozone.

These issues were chosen as priorities for discussion for two reasons: (a) at present the regulatory systems necessary for a better environment are either, not in place or not wholly effective and (b) the issues listed are predicted to be increasing in importance in the next decades in the Netherlands.

Although policies are in place to control land contamination, the steering group decided to include land contamination in the priority assessment in order to see how land contamination compares with the other environmental issues, in terms of priority, for the Netherlands.

There are obvious omissions to this report. Environmental issues not included are; chemical release into air, land and water, waste disposal and the depletion of groundwater. The reason for this exclusion is because these issues are considered to be already regulated and existing targets are expected to be met. In other words, these environmental issues are no longer considered to be the subject of further environmental policy in the Netherlands and as a consequence are not included in the forthcoming NEO5. The issue of biodiversity is not treated as a separate environmental issue because the preservation of biodiversity is a common aim to all the issues covered.

2. Structure of the report

The aim of this research project is twofold:

i) To use benefit estimates as a tool to determine the size of public benefits for environmental issues relevant for the Netherlands, in order to facilitate an environmental priority scheme for the Netherlands; and
ii) To examine the role of cost benefit analysis (CBA) in efficient decision making.

Consequently the report is divided into two parts. The main report is an assessment of the environmental damage due to seven environmental issues in the Netherlands. These results are then interpreted as the primary environmental benefits of pollution control, where benefits are taken as avoided damage. Annex II (*Integrating Cost-Benefit Analysis into the Policy Process)* presents a discussion of CBA as a decision making tool. Part II provides an outline of the structure of the cost benefit approach to environmental policy in particular and policy in general. The advantages of integrating cost-benefit approaches into decision making are discussed as well as some of the controversies surrounding CBA and suggestions are put forward on how they might be resolved. Institutional obstacles to the implementation of CBA are identified and an overview of the ways in which CBA is used in decision making in Europe and the USA is presented.

3. Methodology for setting priorities in environmental policy

3.1 Introduction

Environmental protection is a major concern in the Dutch policy decision making process. However, all measures taken in this area cost money and environmental budgets are limited. In the Netherlands expenditure on pollution abatement[5] is reported at 1.2% of GDP in 1990 (ERECO, 1992). In general, European Union Member States spend an average of 1.1% of their GDPs on pollution abatement (ERECO, 1992). Although these proportions are not fixed through time, substantial increases are not likely in the near future. This suggests efficient use must be made of the economic resources to protect the environment, in other words, environmental expenditure must be cost effective.

Environmental improvement may come as reductions in ambient concentrations of a pollutant, increased land quality or reduced disturbance from noise, etc. The problem for policy is that these gains are measured in different units; such as, micrograms of pollutant per cubic metre of air, micrograms of pollutant per millilitres of water, numbers of people exposed to different noise levels, and so on. A problem of comparability arises and it is not possible to determine whether it is better to spend one more Euro on air quality improvement or noise quality improvement. Monetised values seek to overcome this problem of comparability.

There are three ‘layers’ to the priority setting problem: (a) setting priorities within a given environmental issue, such as air pollution; (b) setting priorities between different environmental issues, such as air pollution versus land contamination control; and, (c) setting priorities between environmental and non-environmental expenditures.

This report is concerned with layers (a) and (b); it does not address (c). The report presents both the methodologies for determining priorities and the rankings that emerge when the methodologies are adapted.

3.2 Overview of methodologies for setting priorities in environmental policy

This chapter presents the methodologies for setting priorities in environmental policy, while the results from applying these methodologies are presented in *Chapters 4, 5* and *6*.

Priority assessment is taken to be in the context of an environmental budget. The underlying methodology required is that of cost-effectiveness, i.e. maximising the benefit to be obtained per Euro of expenditure. The rationale for adopting cost effectiveness as the basic criterion is simple: expenditures that do not maximise effectiveness could have been used for other purposes either within the environmental budget or outside it. Hence, failure to pursue cost effectiveness means that environmental benefits, or some other benefit, such as gains in employment, is being lost for the same expenditure of money.

Unfortunately, information on the costs of implementing environmental policies in the Netherlands or indeed anywhere in the EU, is extremely limited. As a result, the report focuses on the evidence relating to the effectiveness of policy, i.e. the benefits to be obtained. It has to be understood that in the absence of data on the costs of implementing policies, these measures of effectiveness provide

[5] Pollution abatement is defined as the expenditures on abatement of air, water and noise pollution and includes expenditures made by government, industry, household and other organisations.

only part of the picture necessary for establishing priorities. Their primary purpose is one of ‘demonstrating’ the importance of an issue and providing a first approximation of priorities.

Taking cost effectiveness as the basic tool for setting priorities presupposes that there is an agreement on what constitutes ‘effectiveness[6]’. Effectiveness measures can take many forms, those adopted in this report are:

a) monetary damage estimation, i.e. finding the ‘willingness to pay’ (WTP) of individuals for changes in environmental quality and changes in environmental assets, (*Section 3.3*). This indicator underlies the cost benefit analysis approach (discussed in *Part II: Integrating Cost-Benefit Analysis into the Policy Process*);

b) public opinion, i.e. measures of ‘human wellbeing’ based on individual preferences as revealed by public opinion polls (*Section 3.4*); In practice the information on public preferences for environmental policy at the level of detail required for priority setting is extremely limited, and

c) expert opinion, for example, based on the opinions of the steering group for this report. The steering group suggested that the seven environmental issues considered in this study are of general priority for the Netherlands.

It is important to note that three methods used to assess effectiveness use different bases. The monetary damage and public opinion approaches are based on individual preferences, whilst the final method is based on expert opinion. The DALY methodology is also based on expert opinion since different DALYs are weighted by experts (refer to *Section 4.8.2*) We make no argument here as to which is more important. This is an ongoing debate and it is well known that expert and public opinion on environmental risks can diverge widely.

Determining priorities using these approaches is difficult. The main problem is the substantial gaps in knowledge concerning the quantitative scale of environmental damages in both physical and monetary terms and the absence of detailed public opinion research. There is the additional problem due to the absence of detailed assessments of the costs of policy measures. Because of these deficiencies, a judgmental procedure has to be used until better information is generated. Thus the suggested priorities that follow are the result of this judgmental procedure, citing wherever possible the evidence for supposing that issues are or are not of high priority.

3.3 Monetary damage estimation methodology

The methodology used here for the monetary damage approach is similar to that used in the EC study ‘European Environmental Priorities: an Integrated Economic and Environmental Assessment’ (RIVM et al., 2000).

Each environmental issue is assessed separately. The methodology presented is explained in terms of air pollution, although noise follows the same outline. For contaminated land, we make use of land lost, since the Step 2 (pollution to impact) is missing from the analysis, see *Section 4.4* for methodological details.

In general there are five steps necessary for the monetary damage approach. *Figure 1* illustrates the five steps, which are listed below:

[6] ‘Effectiveness’ measures relate to issues, (e.g. air pollution) and can be equated with ‘importance’, whereas ‘cost effectiveness’ tends to refer to an intervention, (e.g. air pollution policy). Thus the importance of an issue, such as local air pollution can be determined by measuring the risks associated with air pollution. The cost effectiveness of measures to control air pollution is directly related to the reduced risks, but also involves a reference to cost and to the potential for other benefits from the intervention besides reduced air pollution.

1. *Inventory of pollutants*: identify pollutants and estimate the tonnes of pollutant emitted, see *Annex 6 on data flows of the 5th National Environmental Outlook*;

2. *Environmental impacts:* identify the environmental impacts and quantify them in physical units by use of dose / exposure response functions where possible;

3. *Monetary values:* estimate the unit cost of the impacts identified above in monetary units;

4. *Monetary damage estimation and aggregation*: estimate mean aggregate monetary value of the environmental impacts for each environmental issue and sum, and

5. *Uncertainty and sensitivity analysis*: test the effects of different assumptions, or possible ranges of values for different pollutants, on the final results.

The notation presented in *Figure 3.1* is as follows:

i	= impact;
j	= pollutant;
b_{ij}	= coefficient linking ambient concentration Aj to a given physical damage;
$STOCK_{ij}$	= stock of receptors at risk of suffering the given damage;
ρA_j	= change in ambient concentration of pollutant j;
P	= price.

Step 1

Current Legislation Scenario
Tonnes of pollutant emissions

Step 2

Impact assessment
Dose/exposure response functions

Bottom-up	**Top-down**
(modelling) $Impact_j = b_j \, . \, STOCKj \, . \, \Delta A_j$	$\Delta A_j = \Delta E_j$

Step 3: bottom up	**Step 3: top down**
Monetary damage per impact Euro / impact	Monetary damage values Euro / unit of pollutant Note that: embedded in this value are: (Euro / impact) x (impact / pollutant) = Euro / pollutant (or bottom-up step 3)

Step 4	**Step 4**
Total monetary damage Euro (known as the cost of impact or benefit of control) Monetary damage = $P_i \, .b_i. \, STOCK_i \, . \, \rho A_i$ Euro = (Euro/impact). (impact/pop.poll).(pop).(poll)	Total monetary damage Euro (known as the cost of impact or benefit of control) Monetary damage = $P_j \, . \, E_j$ Euro = (Euro / tonne) x (tonnes of pollutant)

Step 5

Sensitivity analysis
Confidence intervals

Figure 3.1 The five steps to monetary damage estimation

Step 1: Inventory of pollutants

This step contains the emission levels of the relevant pollutants. RIVM present data for the current status of each environmental issue and the expected future progress given the environmental policies already in place in the Netherlands and the EU. This is referred to as the 'current legislation scenario' (CLS). The data are drawn mainly from the Fifth National Environmental Outlook[7] for the Netherlands (draft report) (final available in July 2000). Data are expressed as tonnes of each pollutant, or exposure levels to different noise bands. All assumptions that are made in reaching the CLS are clearly stated.

Step 2: Environmental impacts

Current monetary valuation literature concentrates on the following environmental impacts for some air pollutants. The impacts considered are analysed in five categories of receptors:

- human health (morbidity and premature mortality);
- buildings and materials;
- crops;
- ecosystems, and
- climate change impacts (other than those included in the above).

Dose / exposure response functions[8] permit the impacts to be quantified and feed into the next stages of the methodology. Dose / exposure response functions (D/ERF) measure the relationship between a unit concentration of a pollutant and its impact on the relevant receptor. With sufficient information about D/ERF it is possible to estimate the size and type of the environmental impact per pollutant, if we know the size of the relevant receptor, e.g. human population. Some assumptions about pollution dispersion are also needed.

The simplest way of using D/ERFs for physical damage assessment is:

$$Physical\ damage = \Sigma b_{ij}\ .\ STOCK_{ij.}.\ \Delta A_j$$

Where i is impact and j is pollutant, b_i is the coefficient linking ambient concentration A_j to a given physical damage, $STOCK_{ij}$ is the stock of receptors at risk of suffering the given damage (e.g. the human population exposed to pollution) and ΔAj is the change in ambient concentration of pollutant j. In all cases, the stock at risk is defined as those receptors actually exposed to the pollutant in question.

There are two ways of using D/ERFs equations; (i) ‘modelling’, otherwise known as ‘bottom-up’ approaches, or (ii) ‘top-down’ approaches. The ‘bottom-up’ approach makes use of D/ERFs directly, they are applied to the stock at risk and emissions data for first hand calculations. Examples of the bottom-up approach in this study are the health impacts of exposusre to PM10 and low level ozone (O_3), where, dose-response functions for different health end points are known for the Netherlands (and at the European level) and these are combined with population at risk data, concentrations of PM10 / O_3 and WTP to avoid each health end point to give total damage to human health due to PM10 / O_3. Thus all variables are measured with the exception of b_i which is usually borrowed from the literature (though it too may be estimated given sufficient time and resources). The extensive EU research programme called ExternE (CEC 1995 and 1997) uses this method. The D/ERFs used in this study are reported in appendices to the relevant chapters.

[7] This report is prepared for the 4th Dutch Environmental Policy Plan, to be published in early 2001.
[8] The term's ‘dose-response’ and ‘exposure-response’ functions are used interchangeably throughout this report. They are used to describe the response to a given exposure of a pollutant in terms of atmospheric concentration, rather than an ingested dose.

The 'top-down' approach makes a simplifying assumption that the relationship between emissions and concentrations is linear and directly proportional. This implies that if emissions of a pollutant increase by x%, the concentration of that pollutant in the area concerned is assumed to increase by x% as well, i.e.

$$\Delta A_j = \Delta E_j$$

Where, Δ is change in, A_j is ambient concentration and E_j is emissions of pollutant *j.* Combining this assumption with that of a constant coefficient b_i (implied in both approaches) implies that physical damages per unit of emissions are assumed constant in the 'top-down' approach. Therefore, it is not necessary to use D/ERFs directly, but it suffices to use the average Euro per unit of pollutant, which was originally estimated using D/ERFs. Adopting the 'top-down' assumption of linearity is generally justifiable, as many validated models are linear, at least for primary pollutants. The 'top-down' method bypasses Step 2 and uses the results of Step 3 directly. Thus we see that the 'top-down'approach uses information and data from other studies that are brought together and applied to the current context for secondary calculation. For purposes of transparency all D/ERFs embedded within the unit damage values are reported at the end of the relevant sections (*Section 4.4 on acidification*) so that estimates may be compared and updated more easily as the literature develops.

Step 3: Monetary values

Following on from Step 2 there are also two ways of using monetary values to measure the damage of pollutants. Since the 'bottom-up' approach measures the physical impact per unit of pollutant, the necessary monetary value is that per physical impact (e.g. WTP to avoid a case of cardiovascular disease). This is a first hand calculation of damage (e.g. Euro / impact). The difficulties with this approach lie in both finding the right estimates for WTP and in estimating the D/ERF. This is because the relationship between emissions and concentrations can vary across sites and over time, as do the receptors or 'stock' exposed to the pollution. For examples of the 'bottum-up' approach see the analysis on particulate matter and low level ozone.

The 'top-down' approach discussed in Step 2 conducts a secondary calculation for damage. The only monetary value the 'top-down' approach requires is the final damage value given in the form of Euro per unit of pollutant. This is because physical and monetary measures of environmental impacts are already embedded within the estimate of damage of Euro per unit of pollution, i.e. Euro / pollutant comes from (Euro / impact) x (impact / pollutant). In order to make use of the 'top-down' approach, it is necessary to take 'bottom-up' step 3 results from other studies. This means the original values are applied outside of the site context where the original study was carried out. Using the results of an original study for another context / site, is called 'benefits transfer'. For further details regarding benefits transfer, refer to the *Annex 2 on Benefits Transfer*.

Due to the time and budget limitations of this study, it has been necessary to rely mainly on the 'top-down' approach to estimate the benefits of environmental control. For example, for air pollutants, we make use of both ExternE (1995 and 1997) and AEA Technology (1999), studies which both follow the 'bottom-up' approach. For each environmental issue, we conduct a literature review of valuation studies in the Netherlands and in the rest of Europe[9] in order to pull out the most reliable 'willingness to pay' (WTP) estimates to avoid environmental damage.

The majority of the studies to be used are conducted outside the Netherlands thus, the 'willingness to pay' (WTP) estimates are adjusted to reflect Dutch WTP values. The rationale for the adjustments to WTP are given below together with an explanation of how it is achieved:

[9] Techniques for the monetary valuation of environmental damage are not reviewed here. Detailed descriptions of methodology can be found in Freeman (1993).

- *Spatial adjustment* (e.g. adjusting UK WTP values to estimate Dutch WTP values): Ready et al (1999) suggest that where impacts are broadly distributed nationally, the nation-wide estimates of purchasing power parity (PPP) adjusted exchange rates provided by OECD are appropriate for this task, and

- *Intertemporal adjustment-past*: (e.g. 1990 values to current 2000 values) by means of the harmonised consumer price index.

For further details of the WTP adjustments, refer to *Annex 2 on Benefits Transfer*.

Although benefits transfer is a practical method that significantly reduces the time and budget requirements for analysis, it should be remembered that the monetary values used in this way contain uncertainties. The main uncertainty is that the validity of making the transfer is not known. This uncertainty differs from the confidence interval relevant to the estimate that is 'borrowed' for the transfer.

It is a priority of this report to be very transparent with regard to the development of the unit damage values. Many assumptions are made and embedded in the damage per unit of emissions values, such as dose / exposure response functions, as mentioned above. For the purposes of transparency, clear and concise statements of assumptions used are reported in the appendices at the end of the relevant sections.

Step 4: Monetary damage estimation and aggregation

Whether the ‘bottom-up’ or ‘top-down’ method is chosen, monetary damage estimation is a simple procedure and follows a typically multiplicative format. For the ‘bottom-up’ method, the relevant units are the concentration of a pollutant (μg/m^3, ppb, ppm). The units for the measurement of human health impacts look like the following:

$$\begin{array}{cccccccccc} \textit{Monetary Damage} & = & \Sigma(P_{ij} & . & B_{ij} & . & STOCK_{ij} & . & \Delta A_{ij}) \\ \textit{Euro} & = & \textit{(Euro/impact)} & . & \textit{(impact/person.}\mu g/m^3\textit{)} & . & \textit{(persons)} & . & (\mu g/m^3) \end{array}$$

Where P_i is the price of or willingness to pay to avoid the impact, and b_{ij}, $STOCK_{ij}$ *and* ΔA_{ij} are as given above.

Whilst for the 'top-down' approach, the aggregation would be as follows:

$$\begin{array}{ccccc} \textit{Monetary Damage} & = & \Sigma(P_j & . & \Delta E_j) \\ \textit{Euro} & = & \textit{(Euro/tonne)} & & \textit{(tonnes of pollutant)} \end{array}$$

Where *Ej* denotes emissions of pollutant *j* and *Pj* is the unit damage value for pollutant *j.* Thus we see that the unit damage values found in Step 3, are combined with the emissions data given in Step 1 to give the aggregate mean monetary value of the environmental impacts in the CLS. Aggregated damage is estimated for 1995, 2010, 2020 and 2030.

The final product of the above process, i.e. Euro damage for each environmental issue can be interpreted as the maximum Euro primary benefit from environmental control, i.e. the avoided damage. Any secondary benefits of control should also be estimated and added to the avoided damage estimates. The total benefits of control can then be compared with the costs of control and contribute to decision making in environmental policy, when such information is available.

Step 5: Uncertainty and sensitivity analysis

The literature on both the physical impacts and monetary values is assessed in terms of its reliability for use in this study. The final monetary damage values for each pollutant presented is subject to uncertainty. Uncertainty arises at every stage of the multiplicative procedure used to estimate the damage values, because each stage introduces an additional parameter. Uncertainty arises for a number of reasons, AEA Technology (1999) suggest the most common are:

- *statistical uncertainty*: those deriving from technical and scientific studies, i.e. dose response functions, the results from valuation studies and data on other variables;
- *model uncertainty*: those arising from benefits transfer procedures i.e. transfer of dose-response functions and valuation results from other countries to the Netherlands;
- *geographical uncertainty*: effects may vary from one location to another. Since damage done tends to be related to the source of the emissions and the sensitivity of the receptor area, there will be uncertainty due to the geographical location of effects;
- *uncertainty about the future*: these derive from assumptions made about future underlying trends in environmental protection;
- *uncertainty about assumptions*: notably the discount rate, and
- *human error*.

The first type can be assessed through statistical methods, giving a confidence interval around a mid estimate. These ranges are reported where possible. Uncertainties of other types are not amenable to quantification, as the probabilities of occurrence, or even the possible future states themselves, may not be known. Formal treatment of such uncertainty is therefore not possible in the same way. However, the key assumptions underlying damage estimates are reported wherever possible.

A distinction must be made between uncertainty that is a bias and uncertainty that is reflected in the wideness of the confidence interval. A bias will occur when, for example, parts of the estimate are omitted, resulting in an underestimate. Whereas if the confidence interval is wide, this suggests that accuracy is low, but it does not imply there is a systematic error in the estimate. Casual commentators argue that the existence of large uncertainties undermines the credibility of the benefits analysis as a tool for policy makers. It is our professional opinion that the presence of large uncertainty makes it all the more essential that benefits analysis is conducted. It serves both to expand knowledge in the area and reduce the uncertainty and it signals to policy makers the potential risks of their actions.

In this study, an indication of the reliability of the mean damage estimates found in step 4 is given by reporting the 68% and 95% confidence intervals wherever possible. Also, where impacts are omitted / included, an indication of the direction of bias in the results is reported.

Issues in valuation: valuing the risk of premature mortality

In the environmental literature there is an ongoing debate on how to value premature mortality. The main differences of view concern whether the relevant value is the WTP to reduce risk at a given time, i.e. the value of statistical life (VOSL) approach, or the WTP to extend 'life-years' by a certain amount, i.e. the value of a life year (VOLY) approach. The former tends to give larger values in the order of Euro 3.47 million in 2000 prices, which have the effect of dominating benefit estimates, compared to hundreds of thousand Euro for the VOLY.

Given that the VOSL / VOLY debate is unresolved, the premature mortality impacts in this study are monetised using the VOSL only. However, there is the issue that air pollution related mortality predominantly affects the elderly (over 85% or premature deaths are in the over 65 group) (Maddison et al., 1997) and there is evidence that values of risk aversion are lower for this age group. Thus the VOSL is adjusted accordingly. Despite its current popularity in some of the environmental literature, the VOLY is not used. It is important to recognise that there are no VOLY estimates of any rigour for Europe. For a more detailed discussion of the issues surrounding the valuation of premature mortality, refer to *Annex 3 on Valuing the Risk of Premature Mortality*.

3.4 Public opinion methodology

This section determines a ranking of environmental issues according to the surveys of Dutch public attitudes towards the environment.

The rationale for turning to public opinion as an assessment procedure is twofold. Firstly, findings from the various European and national surveys show that the environment remains a major concern for the Dutch public. In 1986, 87% and in 1995, 80% see environmental protection as an 'urgent and immediate problem'. Although in 1999 this level drops to 70%, these figures still demonstrate that public opinion about the environment is consistently strong and not greatly influenced by media or political attention.

Secondly, by using public opinion to rank the environmental issues, all Dutch citizens are weighted equally, they are effectively given an equal number of 'votes' on the environment. The final ranking achieved by this assessment is impartial to differences in variables that can affect economic assessments such as income.

Data are taken from the results of seven public opinion surveys, namely, SCP (1993, 1997), IISP (1993) and Eurobarometer (1992, 1995, 1996 and 1999).

The results of the surveys are compared to see if any trends can be identified in Dutch public opinion about the environment.

The original surveys design the question in two main ways:

1) To obtain a percentage of respondents which agree or disagree to a particular statement, and

2) To find the national average of opinion when respondents are asked to rate a particular issue on a sliding scale, for instance, 1-not worried to 4 – very worried.

For the first question type, percentages are taken only from those respondents who answer that they find an issue important, dangerous or serious. The percentages of people who gave these answers are then summed, and a ranking order is presented. Where data are given as a national average, then the averages are simply ranked in order of importance.

Although environmental issues used in the surveys correspond closely with the environmental issues considered in this study, a few difficulties arise, particularly:

Incomplete ranking: Some issues are excluded from surveys, which means rankings are in effect incomplete, for example, land contamination. Exclusion from public opinion surveys does not necessarily indicate they are given a low priority.

Broad categories: Most surveys cite air pollution as an environmental problem. However, reference is not made to the specific pollutants causing the air pollution, such as particulate matter, low level ozone, etc. We assume that 'air pollution' represents both PM_{10} and low level ozone. We also assume that 'excessive use of pesticides, herbicides and fertilisers corresponds with eutrophication as defined in this study.

Over representation: Some environmental issues are expressed as a major concern at more than one level. For example, biodiversity loss is cited at the global, national and local level. In order to capture repeated representation, the average of the percentage values given in the different levels is used.

3.5 Disability adjusted life years' (DALYs)

The DALY methodology largely follows that of Murray and Lopez (1996). They develop the 'disability adjusted life years' measure in order to assess the global disease burden and consequently the health policy priorities in different regions of the world. This health impact measure combines years of life lost and years lived with disease or disability that are weighted according to severity.

In the NEO5 framework adequate data and future projections are available for particulate matter, low level ozone, noise, ultra-violet radiation, radon, home dampness and food borne infectious disease only. For each relevant health outcome we calculate attributable risks by combining population weighted exposure distributions with relative risk estimates derived from the epidemiological literature. Subsequently for each health outcome the number of cases was estimated by combining baseline incidence rates with the attributive risks. Calculations of future disease burden are based on projections of future population structure. We present the set of endpoints used to arrive at estimates of attributable disease burden and the number of DALYs lost. Finally a total exposure attributable disease burden was calculated by aggregating the number of DALYs for each health outcome. The disease burden associated with additional UV-exposure due to ozone layer degradation was calculated by aggregating annual morbidity and mortality estimates of skin cancer and Dutch burden of disease data, (Melse et al., 2000). Statistical uncertainty was assessed using Monte Carlo techniques.

4. Application of damage assessment methodology

The following sections devoted to the seven different environmental issues are organised in the following order. For convenience the corresponding methodological step is indicated.

- brief overview of the environmental issue;
- identification of the sources of emissions and the most widely accepted impacts due to the emissions;
- definition of the scenarios used to represent the current and future status of the environmental issue (Steps 1 and 2);
- summary of the literature review for WTP information, conducted to establish unit damage costs for each pollutant (Step 3);
- estimation of the environmental damage due to each issue (Step 4);
- indication of the level of uncertainty attached to the results (Step 5), and
- appendix (where relevant) with the assumptions embedded in the analysis.

4.1 Climate change

4.1.1 The issue

Emissions of greenhouse gases (GHG) are thought to lead to climate change. The main greenhouse gases are, carbon dioxide (CO_2), nitrous oxide (N_2O) and methane (CH_4). Three other gases are also considered to be GHGs, these are, hexafluorocarbons (HFC), perflurocarbons (PFC) and sulphurhexafluoride (SF6). Whilst still the subject of scientific uncertainty, the Intergovernmental Panel on Climate Change (IPCC, 1995) suggests that a doubling of pre-industrial CO_2 concentrations would produce an aggregate economic loss of 1.5-2.0% of the world GNP. Although some commentators argue that 'waiting' for better scientific evidence is justified, countries are currently signing up to climate change control strategies.

4.1.2 Source of emissions

Carbon dioxide is produced mainly as a result of the combustion of fossil fuels used for electricity generation and transport and in the industrial sector. The major sources of methane are agriculture, power generation, and from anaerobic digestion which takes place in landfill sites and bio-gasification plants. It is worth noting that when methane is collected from waste disposal sites and burnt, it is converted into carbon dioxide and water. Nitrous oxide is produced mainly as a result of industrial activity and agriculture. Other radiatively active gases such as NO_x, SO_2, VOCs and CO are omitted from the review here.

4.1.3 Physical measure of impacts

Greenhouse gases are not considered to cause significant direct damage to the environment or human health. But, as concentrations increase and more heat is trapped in the atmosphere, global warming is expected to occur and consequently affect the climate causing a number of potentially serious impacts, these are listed in *Box 4.1.1*.

Box 4.1.1 Possible climate change impacts

- sea level rise;
- change in agriculture, forests and fisheries;
- change in energy, water, construction, transport and tourism sectors;
- increased risk of disaster: changes in the frequency and severity of storms, floods, droughts, hurricanes and precipitation levels;
- change in biodiversity;
- increased human morbidity and premature mortality, and
- human migration.

Working group 2 of the IPCC is mainly responsible for assessing the impacts of global warming. Scientific research on global warming impacts has largely focused on the case of CO_2 concentration doubling, i.e. the impacts of an atmospheric CO_2 concentration of twice the pre-industrial level ($2xCO_2$). Doubling is an arbitrary benchmark chosen solely for analytical convenience.

There is great uncertainty about the impacts of rising greenhouse gas concentrations on temperature and climate. Issues of non-linearity, i.e. when changes in one variable cause more than proportionate impact on another variable and irreversibility, i.e. changes that once set in motion cannot be reversed, add to the difficulty in measuring the global warming impacts. Also, some impacts may be more dependent on the rate of warming, rather than the absolute level of temperature. Other issues that also add to the complexity of measurement are: regional variation i.e. some regions will suffer from warming whilst others will benefit at least for some sectors, and inertia in the climate system.

This last issue is due to the fact that atmospheric concentrations rather than emissions determine the amount of warming projected by the models. Thus long time lags and the difficulty of detecting climate change increase the difficulty of reliably determining the magnitude and timing of future effects before they begin to occur. Despite these shortcomings, the impacts of global warming are estimated. For a detailed analysis see Bruce et al., (1996).

Current legislation scenario: emissions

The data for the greenhouse gases are taken from the NEO5 (draft report), the ‘medium growth’ scenario (EC) is chosen, thus damage estimates (or maximum primary benefit estimates) are conservative. Results based on emissions data in the ‘high growth’ scenario are interpreted as the ‘worst case scenario’ for the environment, damage estimates are reported in the sensitivity section.

The total emissions for all GHGs from the Netherlands are reported in *Table 4.1.1*. *Table 4.1.1* also gives the sectoral breakdown for CO_2, CH_4 and N_2O.

Table 4.1.1 Greenhouse gas emissions: current legislation scenario

CO_2: (million tonnes)				
	1995	***2010***	***2020***	***2030***
Total	*180.2*	*198.3*	*213.1*	*233.3*
Energy	45.9	47.5	50.0	57.1
Industry	44.3	45.2	49.3	53.7
Transport	32.4	38.7	44.5	52.2
Households	21.8	21.5	20.0	18.3
Refineries	11.5	15.6	17.0	17.8
Services	9.9	13.5	15.3	16.7
Agriculture	9.4	10.4	11.0	11.6
Other	2.5	3.0	3.0	3.0
Waste management	1.5	2.0	2.0	2.0
Construction	0.7	0.8	0.8	0.8
Sewage treatment	0.1	0.1	0.1	0.1
Water	0.0	0.0	0.0	0.0
CH_4: (thousand tonnes)				
Total	*1172*	*703*	*537*	*462*
Waste management	479	217	102	51
Agriculture	479	385	368	343
Energy sector	178	73	38	38
Households	18	16	16	16
Industry	8	5	5	6
Transport	6	3	2	2
Other	5	5	5	5
N_2O: (thousand tonnes)				
Total	71.9	61.6	61.5	63.1
Industry	31.7	32.9	34.2	35.8
Agriculture	27.6	20.9	20.0	19.6
Transport	7.1	2.6	2.2	2.6
Other	5.0	5.0	5.0	5.0
Energy sector	0.4	0.2	0.1	0.1
Total per gas (million tonnes CO_2-eq)				
CO_2	180.2	198.3	213.1	233.3
CH_4	24.6	14.8	11.3	9.7
N_2O	22.3	19.1	19.1	19.6
HFK	6.7	3	5.4	6.5
PFC	2.1	0.9	1.1	1.2
SF6	1.5	2.0	2.3	2.7
Total	*237.3*	*238.1*	*252.2*	*273.0*

Source: NEO5 (draft report), EC (medium growth).

4.1.4 Monetary measure of impact

Unit damage values for carbon

There are a number of studies, which assess the monetised value of damage done from global warming, these are summarised in *Table 4.1.2*. As a consequence of the scientific focus, the studies tend to be based on the 2xCO_2 scenario as well. The most studied aspects are the impacts on agriculture and the costs of sea level rise, with some studies on forestry. Several studies provide a first order assessment of the total global warming damage using a simple enumerative approach, i.e. the total damage is the sum of individual damage categories. This type of partial equilibrium approach means that the estimates do not include the feedback effects (e.g. the change in agricultural yields will induce changes on the food, tobacco and textile industry). All of the economic assessments undertaken to date are incomplete as they exclude some impacts which are potentially important, including low-risk, high consequence events which are problematic to quantify and value. Despite such short-comings, professional judgement suggests that global warming impacts are recommended for monetary evaluation.

The relevant damage estimates are *marginal damage*, i.e. the damage done by increasing greenhouse gas emissions now by a small amount. Computing marginal damage is not, however, straightforward (for an outline interpretation, refer to the *Appendix* at the end of this chapter, based on Fankhauser (1995)). The relevant concept is the present value of damage done from an increase in emissions, allowing for the fact that greenhouse gases are cumulative pollutants, i.e. an increase in emissions now resides in the atmosphere for a long period of time. Thus,

$$PV(D) = \Sigma \Delta D_t / \Delta E.(1+s)^{-t}, \quad t = 1...T$$

Where PV is present value, Δ is change in, D is damage, E is emissions, s is the discount rate, t is time, and T is time horizon. The complexities involve:

- relating emissions to concentrations;
- relating concentrations to radiative forcing and warming;
- relating warming to damage (since the ratio of damages per tonne of gas is not the same as global warming potentials - see the guide to climate change models at the end of this chapter), and
- selecting the discount rate.

Two forms of estimate are produced in the literature: (i) estimates of actual damage arising at the point in time when $2xCO_2$ occurs and arising from a small increase in emissions now, and (ii) a ‘shadow price’ defined as the level of tax required to keep emissions on an optimal trajectory as estimated by the modeller. Because the estimates relate to damage or benefit at the time when $2xCO_2$ occurs, they are then discounted back to the present (see formula above) so that the choice of discount rate matters. Indeed, it is the discount rate that largely accounts for the difference in estimates.

The results of a literature review of recent studies which assess the economic costs of global warming are summarised in *Table 4.1.2*. Note that the estimates of marginal damage costs from global warming given in *Table 4.1.2* are for carbon only.

Table 4.1.2 Estimates of marginal damage costs from global warming[10]

	Carbon only, Euro / tC. Base year prices: 2000			
Period	*1991-2000*	*2001-2010*	*2011-2020*	*2021-2030*
Nordhau, 1991	9.3	9.3	-	-
Nordhaus, 1994				
P=0.03, best guess	6.8	8.7	11.0	12.8
P=0.03, expected value	15.4	23.0	33.9	-
Nordhaus, 1998	6.4	9.1	11.9	15.0
Fankhauser, 1995				
With p =0,0.005,0.03 (random)	25.3	27.5	31.6	34.8
With p =0	62.5	-	-	80.5
With p =0.03	7.0	-	-	10.6
Cline 1993, with s = 0	7.4 - 158.7	9.7 - 197.1	12.5 - 238.1	15.1 - 282.9
Peck and Teisberg, 1993				
with p =0.03	12.8 - 15.4	15.4 - 17.9	17.9 - 23.0	23.0 - 28.2
Maddison, 1994	7.6 - 7.8	10.4 - 10.8	14.2 - 14.8	18.8 - 19.4
Eyre et al., 1997				
with s = 0	181.8	190.7		
with s = 1	93.4	92.2		
with s = 3	29.4	25.6		
with s = 5	11.5	9.0		
with s = 10	2.6	1.3		
Tol, 1999	14.1	16.6	19.2	23.0
Roughgarden and Schneider, 1999: Lower bound = Nordhaus, Upper bound = Tol	6.4 - 14.1	7.7 - 16.6	10.2 - 20.5	12.8 - 26.9

Note: original estimates in 1990 US$, we assume an escalation of 2.5% p.a. inflation to obtain 2000$ (i.e. a deflator of 1.25), and an exchange rate of US$1 = Euro 1 to get Euro 2000 values.

Eyre et al estimates are for 1995-2004 and 2005-2014 and the estimates here exclude equity weighting. p= utility discount rate and s = the overall discount rate. Roughgarden and Schneider's ranges derive from placing the models of Fankhauser (1995), Cline (1992), Titus (1992) and Tol (1995) into Nordhaus's DICE model framework. The upper end of the range should, strictly, coincide with the marginal damage estimates in Tol (1999).

Table 4.1.2 shows the considerable sensitivity of estimates to discount rates. The discount rates given for Fankhauser's estimates relate to the pure time preference rate component, p, only. According to

[10] Otherwise referred to as the marginal primary benefits from optimal control of global warming.

Fankhauser (1995) his social cost estimates based on the distribution of values for p are equivalent to a 'best guess' value of 0.5% for p. To this must be added a value for the elasticity of the marginal utility of income multiplied by the expected growth rate. Fankhauser and Eyre et al take the elasticity to be about unity, so that the only variable is the expected long term economic growth rate of income per capita. Rabl (1996) suggests this is 1.6-1.8% pa, so that the discount rate would be 2.2 to 2.3%. Accordingly, the discount rate value in Eyre etc of 3% is more relevant for purposes of comparison.

Several factors explain the variability of the estimates in *Table 4.1.2*. First, the effect of different discount rates is shown by the values of 'p', the utility discount rate. This rate discounts future utility and it is usual to add this rate to the rate for discounting future income (consumption). Controversy surrounds the value of 'p' since some authors regard utility discounting as illicit, i.e. they set p=0. The effect is easily seen by comparing Fankhauser's estimates with utility discount rates of zero and 3%: the difference is a factor of 9 in the damage estimate. The discount rate partly explains Cline's large range, but his estimates also reflect very high estimates of damage and a very long-term time horizon. The 'central' estimates are surprisingly similar. For 1991-2000 the damage estimate is around Euro7-15 tC but with the Eyre et al estimates being around twice this. Interestingly, the most recent studies, by Tol and Nordhaus suggest damage estimates lower than those previously estimated, although Nordhaus's estimate is fairly constant at Euro 6-9 (taking his preferred 'best guess'). Tol's 1999 paper is also probably the most careful recent estimate. Fankhauser's model has considerable attractions because of its use of the discount rate as a random variable. This is shown here in the row with p=0, 0.005, 0.03, i.e. with a probability distribution assumed for 'p' taking values of 0%, 0.5% and 3%.

Roughgarden and Schneider (1999) take four studies of *total* damages for $2xCO_2$ concentrations: Titus (1992), Cline (1992), Fankhauser (1995) and Tol (1995). Making slight modifications to the damage estimates they then derive *damage functions* of the form:

$$D_t = a[\Delta T_t/w]^2$$

Where D_t is the monetary value of damage done in year t expressed as a fraction of gross world output, a is the fraction of world GNP lost for $2xCO_2$, ΔT_t is the change in earth surface temperature, w is the expected temperature change in ^oC for a $2xCO_2$ benchmark. Thus, in Tol's case, for example, a = 0.02, i.e. $2xCO_2$ is expected to give rise to global output loss of 2%, and w is 2.5. These damage functions are put into Nordhaus's DICE model to produce optimal carbon taxes, (i.e. marginal damage estimates) as shown in the final row of *Table 4.1.2.*

Recent work by Mendelsohn and Neumann (1999) suggests *net benefits* for impacts on the market sector in the USA, and Mendelsohn (1996) has also suggested that this conclusion may hold true for the world as a whole. This is not the position taken here. Also ignored is the effect of equity weighting on the estimates. For this debate see Fankhauser et al., (1997a, 1997b), Tol et al., (1996, 1999), Azar (1999), and Azar and Sterner (1996).

The authors acknowledge that the discount rates assumed in the economic studies of climate change will have a significant impact on the results. Where marginal damage values are present values over the entire period, we assume discount rates of zero too low and 10 too high. Thus, marginal damages of Euro 8 - 30 tC for the year 2000 and onwards seem defensible as a central range. The unit damage values assume damage to drylands, wetlands, ecosystems, agriculture, forestry, energy, water sector, amenity, human health and coastal regions.

The reliability of the recommended central marginal damage values is found using the confidence limits around these mean marginal damages. The upper value of the recommended central estimate, Euro 30/tC, is very close to the estimates given by Fankhauser's model (1995). Based on the 90% confidence limit for his marginal damage values, we present a range of marginal damage values around the upper central value at Euro 9 - 67 pt C. Due to information limitations we are not able to provide a range around the lower central estimate.

The marginal damage values for carbon used in this study are summarised in *Table 4.1.3*.

Table 4.1.3 Marginal damage values for carbon: Euro (2000 values) per tonne

Year	Mean	90% confidence interval for upper mean value Fankhauser (1995)
2000 onwards	8 - 30	9 - 67

Unit damage values for other GHGs

Finally we require values for the other greenhouse gases. A common procedure is to use the global warming potential (GWP) as a mechanism for expressing all greenhouse gases as carbon dioxide equivalents. We take GWP ratios as follows:

CO_2	1
CH_4	21
N_2O	310

Thus, emissions of methane and nitrous oxide are converted to CO_2 equivalent (using GWP ratios), then converted into tonnes of carbon i.e. multiplied by 12/44. The tonnes of carbon are summed across the three pollutants and valued using the marginal damage values per tonne of carbon given in *Table 4.1.3*.

However, as Fankhauser (1995) and others have pointed out, GWPs are not necessarily the appropriate multiplier if the focus is on damage. For GWPs to be the correct adjustment, damage would have to be linearly related to radiative forcing. But the two are not linearly related since damage depends on other factors as well, such as the previous level of radiative forcing and the level of warming that has already taken place. Fankhauser (1995) estimates the marginal damages for the main greenhouse gases as shown in *Table 4.1.4*. Eyre (1997) suggests very similar values to Fankhauser for both nitrous oxide and methane.

Table 4.1.4 Marginal damage values for the main greenhouse gases: Euro / t pollutant, (Euro 2000)

Greenhouse gas	Marginal damage values in different year spans			
	1991 - 2000	2001 - 2010	2011 - 2020	2021 - 2030
CH_4 As Euro / tCH_4				
Fankhauser (1995)	135.0 (60.0 - 256.3)	161.3 (72.5 - 311.3)	190.0 (86.3 - 366.3)	220.0 (97.7 - 427.5)
Eyre et al. (1997)	127.0	149.8		
N_2Oas Euro / tN				
Fankhauser (1995)	3618 (1006 - 9066)	4223 (1191 - 10451)	4876 (1402 - 12101)	5611 (1596 - 13901)
Eyre et al. (1997)	4751	4752		

Note: original estimates are in 1990 US$, we assume an escalation of 2.5% p.a. inflation to obtain 2000$ (i.e. an inflator of 1.25), and an exchange rate of US$1 = Euro 1 to get Euro 2000 values.

Secondary benefits

It is important to distinguish between the damages of global warming and the benefits of climate control policies. The benefits include the avoided damages but are likely to be significantly greater because of the ancillary gains from climate control policies, the so-called 'secondary benefits'.

Secondary benefits arise because the control of carbon dioxide is likely to involve policies, which will also reduce other pollutants. Most of the literature has concentrated on the effect of climate control policies on jointly produced pollutants such as NO_x, SO_x etc. However, it can be argued that climate change policies may also secure gains in employment and in other social effects such as reduced noise nuisance and accidents. How far gains in employment should be credited to climate policies, however, is open to debate. If such gains could be secured anyway, i.e. independently of climate change policy, then it would not appear to be reasonable to add them in as benefits of a climate policy. If they can only be secured by a climate policy, then their addition seems appropriate. Thus the total benefits i.e. primary and secondary benefits of a policy measure to control greenhouse gases are likely to exceed the avoided damage i.e. primary benefits of control.

Table 4.1.5 reveals a wide disparity of estimates. US estimates are, with some exceptions, notably smaller than European estimates. Part of the problem arises from slightly different methodologies: the European studies tend to be 'snapshot' pictures of the amount (and value) of secondary pollutants emitted at any one time. These are then expressed per tonne of carbon. However, for policy purposes it is preferable to estimate such benefits as being incremental to any policies currently 'in the pipeline' for secondary pollution policy. Since most secondary pollutants are the subject of independent policies, emissions can be expected to fall over time, so that climate change policies will secure a further but smaller incremental fall in the future.

The evaluation of the climate change issue presented in this report relates to current damage in 1995 and future damage in 2010, 2020 and 2030. This type of analysis makes use of the marginal damage values reported earlier, it does not require the inclusion of the secondary benefit estimates reported in *Table 4.1.5*. However, when evaluation of global warming control policies is conducted, it is important to consider the secondary benefits of those policies.

Table 4.1.5 Secondary (emission) benefits per tC as a multiple of primary benefits

Study	Country	Ancillary benefits Euro / tC	Ancillary benefits as a multiple of primary benefits (at Euro 30 / tC)	Comment
Ayres and Walter, 1991	USA	165	5.5	
Boyd et al., 1995	USA	40	1.3	Criteria pollutants
Burtraw and Toman, 1998	USA	<10	<0.33	Judgmental assessment of prior studies
Burtraw et al.,1999	USA	3	0.1	SO2, NOx only
Dowlatabadi et al.,1993	USA	3	0.1	SO2, NOx, PM
Goulder, 1993; Scheraga and Leary, 1993	USA	33	1.1	SO2, NOx, PM, Pb, CO, VOCs
Lutter and Shogren, 1999	USA	300	10.0	See text
Rowe et al., 1995	USA	24	0.8	SO2, NOx, PM
Viscusi et al., 1994	USA	88	2.9	Criteria pollutants
Barker, 1993	UK	44 - 201	1.5-6.7	Relies on Pearce 1992
Pearce, 1992	UK	195	6.5	SO2, NOx, PM
Ayres and Walter, 1991	Germany	312	10.4	
Alfsen, 1992	Norway	102 - 146	3.4-4.9	
RIVM et al., 2000	EU	53 - 79	1.8-2.6	General equilibrium model

Sources: OECD (1999); Lutter and Shogren (1999); RIVM et al., (2000).

4.1.5 Aggregate monetary damage estimate

Due to the existence of an international climate change treaty the relevant estimate is damage to the world caused by greenhouse gas emissions from the Netherlands.

The following tables present the results of applying the shadow values in *Table 4.1.3* to the current legislation scenario (CLS) emissions for the greenhouse gases given in *Table 4.1.1*. The method used converts all greenhouse gases into carbon equivalents, using the global warming potentials. *Table 4.1.6* reports the mean damage to the world due to Dutch GHG emissions in the years 1995, 2010, 2020 and 2030. All values are given in millions of Euro (2000 prices).

Table 4.1.6 Mean damage to world due to Dutch GHG emissions: Euro million

	1995	2010	2020	2030
CO_2 as C	393 - 1474	433 - 1622	465 - 1743	509 – 1908
CH_4 as Ceq	54 - 201	32 - 121	25 - 92	21 – 79
N_2O as Ceq	49 - 182	42 - 156	42 - 156	43 – 160
Subtotal	***495 - 1858***	***507 - 1900***	***531 - 1991***	***573 – 2148***
HFC as Ceq	15 - 55	7 - 26	12 - 44	14 – 53
PFC as Ceq	5 - 17	2 - 7	2 - 9	3 – 10
SF6 as Ceq	3 - 12	4 - 16	5 - 19	6 – 22
Total	**518 - 1942**	**519 - 1948**	**550 - 2064**	**596 – 2234**

Note: the range of damage estimates presented in *Table 4.1.6* are based on the low - upper central marginal damage values, i.e. Euro 8 - 30 /tC.

Table 4.1.7 presents the mean damage to the world due to the emissions of CO_2, CH_4 and N_2O from the Netherlands by sector, in the years, 1995, 2000, 2010 and 2020. Sectoral emission data for the other GHGs are not available. The greatest level of damage is caused by carbon dioxide emissions from the power generation, industrial and transport sectors. One of the most striking changes in damage between the years 1995 and 2020 is the level of damage caused by methane emissions from the waste sector which goes down significantly.

Table 4.1.7 Sectoral mean damage due to CO_2, CH_4 and N_2O. Euro million (2000 prices)

CO_2 as C				
	1995	2010	2020	2030
Waste management	12.6	16.1	16.1	16.1
Construction	5.8	6.6	6.6	6.4
Households	178.6	176.1	163.9	149.6
Water	0.2	0.0	0.0	0.0
Energy	375.4	388.6	409.1	466.8
Services	81.4	110.5	124.8	136.8
Industry	362.2	369.5	403.4	439.3
Agriculture	76.8	84.9	90.0	95.3
Other	20.4	24.5	24.5	24.5
Refineries	94.1	127.9	139.2	145.7
Sewage treatment	1.0	1.2	1.2	1.2
Transport	265.4	316.5	364.4	427.2
Total	***1474.0***	***1622.4***	***1743.3***	***1908.8***
CH_4 as Ceq				
Waste management	82.3	37.3	17.6	8.8
Households	3.0	2.8	2.8	2.8
Energy sector	30.6	12.5	6.5	6.5
Industry	1.3	0.8	0.9	1.0
Agriculture	82.2	66.2	63.3	59.0
Other	0.9	0.8	0.8	0.8
Transport	1.0	0.5	0.4	0.4
Total	***201.3***	***120.9***	***92.2***	***79.3***
N_2O as Ceq				
Energy sector	1.0	0.5	0.3	0.3
Industry	80.4	83.3	86.6	90.7
Agriculture	70.0	53.0	50.8	49.7
Other	12.8	12.6	12.6	12.6
Transport	18.1	6.7	5.6	6.7
Total	***182.3***	***156.2***	***155.9***	***160.1***

Note: for clarity damage estimates presented in *Table 4.1.7* are based on the upper central unit damage value for a tonne of C only, i.e. Euro 30/tC.

Finally, *Table 4.1.8* gives the mean damage to the world due to Dutch emissions of the main greenhouse gases by sector. The results show that between the years 1995 to 2020, the sectors that contribute most to the climate change problem from the Netherlands are, in descending order: industry, power generation and transport followed by agriculture. The order is held throughout the entire period. Although, the percentage share of environmental damage due to the industrial sector does increase slightly from 23.8% to 24.8%, whilst the power generation sector maintains a constant share of 22.0%. The transport sector increases from 15.3% to 20.2% and the agriculture sector maintains a constant share of 12%.

Table 4.1.8 Sectoral mean damage due to CO_2, CH_4 and N_2O emissions: Euro million

	1995	2010	2020	2030
Industry	118 - 444	121 - 454	131 - 491	142 - 531
Energy	109 - 407	107 - 402	111 - 416	126 - 474
Transport	76 - 285	86 - 324	99 - 370	116 - 434
Agriculture	61 - 229	54 - 204	54 - 204	54 - 204
Households	48 - 182	48 - 179	44 - 167	41 - 152
Waste management	25 - 95	14 - 53	9 - 34	7 - 25
Refineries	25 - 94	34 - 128	37 - 139	39 - 146
Services	22 - 81	29 - 111	33 - 125	36 - 137
Other	9 - 34	10 - 38	10 - 38	10 - 38
Construction	2 - 6	2 - 7	2 - 7	2 - 6
Sewage treatment	0 - 1	0 - 1	0 - 1	0 - 1
Total	**495 - 1858**	**507 - 1900**	**531 - 1991**	**573 - 2148**

Note: the range of damage estimates presented in *Table 4.1.8* are based on low - upper marginal damage values for carbon, i.e. Euro 8 - 30/tC. Sectoral emissions data for HFC, PFC and SF6 are not estimated, thus the damage estimates due to these gases are excluded from *Table 4.1.8*.

4.1.6 Uncertainty

The extent of uncertainty for the damage estimates is large. The main areas of uncertainty are climatological, i.e. the expected changes in global mean temperature and changes in temperature dependent factors such as sea level, precipitation, evaporation etc. The uncertainty continues in the estimation of the impacts to ecosystems, agricultural yields and human health. The monetised environmental damage estimates presented in this section are likely to be biased. The reasons for the possible downward and upward bias are as follows;

- damage estimates are calculated using the marginal damage values, which include damage to drylands, wetlands, ecosystems, agriculture, forestry, energy, water sector, amenity, human health and coastal regions. The omission of damage to other sectors dependent on climate, such as tourism, transport, construction and insurance suggest that the damage estimates will be biased downwards;

- damage estimates are based on the GWP approach, the GWP ratios are assumed to be C: 1, CH_4: 21 and N_2O: 310. GWPs are not necessarily the appropriate multiplier if the focus is on damage. Based on the work of Fankhauser (1995) we derive 'Global Damage Potential' (GDP) ratios, these are, C: 1, CH_4, 20.7 and N_2O (see *Appendix* at the end of this chapter): 172.9. This suggests the damage estimates based on the GWP approach may be overstated;

- damage estimates may be overstated due to the omission of adaptation strategies (Mendelsohn and Neumann, 1999) in the earlier studies estimating damage, such as farming adaptation strategies, and

- total benefits of climate change control policies will be greater than the damage avoided (primary benefits) if the secondary benefits to other environmental issues are included.

Sensitivity analysis

Assumptions are made throughout this analysis. Some may have a significant effect on the results, while others will only make a minor difference. This section examines what happens to the damage estimates if the assumptions are changed.

The sensitivities considered are as follows:

- effect of using the high growth scenario for greenhouse gas emissions,
- effect of using the GDP approach to damage assessment, and
- assume high discount rate, s=10%.

We also present estimates based on lower and upper marginal damage values. The results of the sensitivities are presented in *Table 4.1.9*.

Table 4.1.9 *Sensitivity test 1: Mean, lower, upper damage caused by GHGs: Euro million*

	1995	2010	2020	2030
Current legislation scenario estimate GWP approach: mean marginal unit damage value: Euro 30/tC	1942	1948	2064	2234
Sensitivities (i) lower and upper values around the mean, Euro 30/tC, Euro 9 - 67/tC respectively; (ii) high growth scenario; (iii) GDP approach: mean marginal unit damage values per tC, per tCH_4 and per tN; (iv) GDP approach: lower and upper values; (v) high discount rate, s= 10%, i.e. marginal damage values becomes Euro 1.9 / tC	583 - 4337 1942 1488 474 - 3284 124	584 - 4350 2034 1738 575 - 4000 126	619 - 4609 2219 2035 677 - 4675 132	670 - 4988 2502 2425 809 - 5582 141

From the sensitivity analysis, we can conclude the following:

- the mean marginal damage value used in this study, (Euro 30 /tC) is supported by the work of Fankhauser (1995). The reliability of these values is measured by presenting damage estimates based on the lower and upper 90% confidence limit. *Table 4.1.9* shows that the damage estimates for climate change can be estimated to within a factor of roughly 2 (90% confidence interval). Thus, the damage estimates presented should be interpreted as a rough assessment of the order of magnitude only.

- estimates based on the high growth scenario for greenhouse gas emissions (Fifth National Environmental Outlook, draft) may be interpreted as a 'worst case scenario for the environment. The damage estimates are greater than for the medium growth scenario by 4% in 2010, 7% in 2020 and 11% in 2030.

- in the main analysis we assume all greenhouse gases are converted to carbon equivalents, by using their respective GWP ratios. Here we compare these results with the results derived from valuing the individual GHGs with their respective marginal damage value, i.e. Euro / tC, Euro / tCH_4, and Euro / tN. Note carbon dioxide is converted into tonnes of carbon, and nitrous oxide is converted into tonnes of nitrogen, by using their respective, relative molecular weights. Changing the approach to valuing GHGs, i.e. the GWP approach: Euro / tC for all GHGs, or using the GDP approach: Euro / t pollutants for the different GHGs, makes very little difference to the outcome. For example, comparing the mean results in 1995, the GWP approach, suggests environmental damage is some Euro 1.9 billion, whilst using the GDP approach the results are given as Euro 1.5 billion. However, over time the damage estimates based on the GDP approach become greater than the damage estimates based on the GWP approach. This is explained by the rising marginal damage values per pollutant over time, see *Table 4.1.10*, compared to the constant marginal damage values assumed in the GWP approach, i.e. Euro 30/tC. Estimates based on the GDP approach are an underestimate because they assume damage caused by emissions of CO_2, CH_4 and N_2O only.

Table 4.1.10 Adjusted marginal damage values used in the GDP approach: Euro 2000

	1991 - 2000	2001 - 10	2011 - 20	2021 - 30
C: mean Euro / tC	25.4	28.5	31.6	34.8
CH_4: mean Euro / tCH_4	135.0	161.3	190.0	220.0
N: mean Euro / tN	3618.8	4223.8	4876.3	5611.3

Note: original values adjusted by 2.5% p.a. i.e. the inflator 1.25, Financial exchange rate in Jan 2000: US$ = Euro 1.

- the final sensitivity test challenges the discount rate assumption. Assumptions about discount rates will have a major impact on the final results. There is no consensus on which discount rate to use. (Thus it is possible to choose either to not monetise damages or monetise climate change damages using a large range of values). It is our professional opinion that discount rates of zero are unrealistic and 10% is high. In the main analysis we assume a discount rate between 0-10. In the sensitivity we present results based the discount rate 10%.

Appendix Guide to economic models of climate change damage

Economic damage estimates: underlying models

Assumption: The rise in sea levels connected with a warmer climate will threaten low lying coastal regions. Sea level rise will particularly affect densely populated coastlines, and small island states. Most studies agree, in principle, that damage estimates should be based on the assumption of a cost efficient response. That is, damage should be kept at a minimum through appropriate adaptation measures. This might include the erection of sea defences, the development of heat resistant crops, changes in agricultural practice and forest management, the construction of water storage and irrigation systems, the adaptation of houses, etc (Fankhauser, 1995). Unfortunately, while undisputed as a general idea, data limitations do not always permit the implementation of optimal adaptation strategies when it comes to actual damage calculations. Thus it is assumed, for simplicity that no adjustments are taken at all, or that current service levels are maintained.

The underlying models for the economic damage estimates are fairly complex. The following is an outline interpretation.

The models deal with small ('marginal') increases in emissions in a base period (e.g. now). This leads to an increase in atmospheric concentrations

$\partial C_i(s)/\partial E(0)$

where s is the time of the concentration, 0 is the base year of emission.

In turn, changed concentrations give rise to changed temperatures which depend on the radiative forcing of the gas, concentrations of other gases, climatic feedbacks and the inertia of the climate system:

$\partial T(t)/\partial C_i(s)$, for $t>s$.

Increased temperatures give rise to (marginal) damage

$\partial D(t)/\partial T(t)$.

Taking all the links together, the incremental damage done by a marginal emission *at time t* now will be:

$\partial D(t)/\partial E_i(0) = \partial D(t)/\partial T(t).\int_{0,t} \{\partial T(t)/\partial C_i(s).\partial C_i(s)/\partial E(0)\}\ ds$

The marginal damage costs of an incremental tonne of emissions are

$K_i = \int_{0,\tau}[\partial D(t)/\partial E_i(0)].e^{-rt}.dt$

since damage done is cumulative (i.e. occurs as long as the gas remains in the atmosphere) and needs to be discounted back to the present. r is the discount rate.

So,

$K_i = \int_{0,\tau}[\partial D(t)/\partial T(t).\int_{0,t} \{\partial T(t)/\partial C_i(s).\partial C_i(s)/\partial E(0)\}\ ds]e^{-rt}.dt$

Damage and Radiative Forcing

The ratio of marginal damage from gas i, relative to CO_2, will be the same as the ratio of GWPs if and only if damage is a linear function of radiative forcing. But this is not true, since the damage from a given increase in radiative forcing will depend on previous levels of forcing and the degree of warming already encountered.

As an example, Fankhauser (1995) has an *annual* damage function taking the form:

$D_t = k_t \{T_t/\Lambda)^{\gamma}.(1+ø)^{t^*-t}$

where T in this case is surface temperature, Λ is the amount of warming associated with a doubling of CO_2 (in Fankhauser's case, 2.5°), t* is the time this is expected to occur (2050), k_t is the economic damage done by $2xCO_2$. Parameter γ is the relationship between temperature and damage, i.e. if temperature rises by 1%, damage rises by γ%. ø is a factor which makes impacts greater if they occur before t* and lower if they occur after t* - a rough attempt to account for damage being related to speed of change.

Thus, if $2xCO_2$ damage occurs in 2050 then damage = k_t since t*=t and T_t = 2.5°C. Fankhauser (1995) adopts the following values based on the scientific and economic literature:

Λ = 2.5°
t* = 2050
γ = range 1 to 3, with best guess 1.3
ø is random with best guess of 0.006
K_t is 'bottom up' procedure of aggregating individual damages. Damages are adjusted for population and economic growth.

In any period t, then, annual damage is given by

$D_t = \$270.10^9.(T_t/2.5)^{1.3}$

The model is not linear. Atmospheric concentrations are linear with respect to emissions; forcing is logarithmic in CO_2 and has a quadratic form for CH_4 and N_20 (following IPCC); temperature change is linear with forcing; and damages are not linear with temperature. Thus use of forcing ratios to measure the *relative contributions* of different gases to *damage* is misleading. An illustration is taken from Fankhauser (1995). The latest GWPs and global damage potentials are reported in *Table 4.1.11*.

Table 4.1.11 Global warming potential and global damage potential ratios

Gas	Global warming potential ratios (100 years)	Global damage potential ratios Fankhauser (1995)
CO_2	1	1
CH_4	21	20.7
N_2O	310	172.9

Note; to estimate global damage potentials, i) take marginal damage values for C, CH_4 and N given in *Table 4.1.3* and *Table 4.1.5,* for the years 2001 - 2010; i.e. Euro 28.5/tC, Euro 161.3/ tCH_4 and Euro 4223.8/tN; ii) convert to marginal damage values for a tonne of CO_2, CH_4 and N_2O, i.e. Euro 7.8/tCO_2, Euro 161.3/tCH_4 and Euro 1343.9/tN_2O; iii) estimate global damage potential ratios, i.e. divide marginal damage values for CO_2 into values for CH_4 and N_2O.

4.2 Acidification

4.2.1 The issue

The primary pollutants, sulphur dioxide (SO_2), nitrogen oxides (NO_x) and ammonia (NH_3) together with their reaction products lead, after their deposition, to changes in the chemical composition of the 'acidification'. Their concentration in the air can cause impacts to human health.

4.2.2 Source of emissions

The largest source of anthropogenic nitrogen oxides is the transport sector. This accounts for 63% of all NO_x emissions in the Netherlands. Other important sources are industry (12%) and the energy sector (12%). Sulphur dioxide is produced mainly as a result of industrial activity, i.e. a by-product of power generation (12%), refineries (43%) and manufacturing (21%). The contribution of transport is again significant (20%). By far the largest source of atmospheric ammonia is agricultural activity (94%) with consumers, industry and power generation contributing very small amounts.

4.2.3 Physical measure of impacts

There are five major receptors of air pollution: human health, crops, materials damage, ecosystem damage and visibility impairment. However only the initial three can be quantified with any certainty at this stage. Thus overall damages across the three receptors are estimated and given as the likely benefits of the reduction in emissions. *Table 4.2.1* outlines the main receptors of damage from acidifying emissions, providing a brief description of each impact and the pollutant responsible. *Table 4.2.1* also describes those impacts omitted from economic valuation, due to either the current scientific or economic uncertainty.

Note that the effects of ammonia are covered under other pollutants i.e. aerosols, N deposition, acidic deposition. Impacts included are health (morbidity and premature mortality), fertilisation impact on agriculture. Impacts to ecosystems are excluded. However for eutrophication impacts, refer to *Chapter 4.6*.

Recommended D/ERFs are given in full in the *Appendix* to this chapter. If it is assumed to be a linear relationship, then for simplicity, only the D/ERF coefficient *b*, is reported. Unless described otherwise, *b,* should be interpreted as the increase in annual incidence of each impact. For example, for morbidity, *b* is the number of cases per $\mu g/m^3$ per year over a given population and for mortality, *b* is the % change in mortality rate per $\mu g/m^3$ per year.

Table 4.2.1 Environmental impacts of SO_x, NO_x and NH_3

Impact receptor	SO_x	NO_x
Impacts quantified		
Human health	Direct damage Indirect damage through sulphates (SO_4 and SO_2 aerosols)	Evidence of direct damage is mixed Indirect damage through nitrates (NO_3 aerosols) and ozone, (O_3)
Crops	Fertilisation effect at low concentrations Damage at high concentrations	Fertilisation effect (N deposition from NOx and NH_3)
Materials	Damage (SO_2)	No effect
Impacts not quantified		
Ecosystem	Damage	Not known
Visibility	Possible reduced visibility due to increased air pollution	Possible reduced visibility due to increased air pollution
Materials	Effects on cultural assets (SO_2) Damage to steel in reinforced concrete (SO_2)	No effect
Agriculture	Indirect effects on livestock (SO_2)	No effect
Human health	No effect	Indirect damage through ozone, (O_3): see *low level ozone* for analysis.
Forests	No effect	Maybe some beneficial effect
Crops	All pollutants: Interactions between pollutants, with pests, pathogens and climate (i.e. the negative impact of SO_2 on global warming is omitted	
Ecosystems	Excess N deposition (from NOx and NH_3 emissions) may harm ecosystems through the effects of eutrophication.	

The analysis here follows the 'top-down' approach described in the methodology (see *Section 3.3*). This means that D/ERFs are not used directly but are embedded within the unit damage values for the acidifying pollutants. For details of the D/ERFs used to establish unit damage values (and other assumptions, such as the 'stock at risk'), refer to the *Appendix* at the end of this chapter.

Current legislation scenario: emissions

The data for the acidifying pollutants are drawn from the NEO5 (draft report), using the medium scenario ('EC'). Total emissions for SO_2, NO_x and NH_3 from the Netherlands are reported in *Table 4.2.2*. *Table 4.2.2* also gives the sectoral breakdown of emissions sources.

Table 4.2.2 Gases contributing to acidification: current legislation scenario

Year	1995	2010	2020	2030
SO_2: 1000 tonnes				
Waste	0.5	0.3	0.3	0.3
Energy	16.7	9.8	8.6	10.5
Industry	29.8	23.8	24.7	25.7
Other	5.7	2.7	2.8	2.8
Refineries	61.2	21.0	22.0	22.3
Transport	29.9	13.7	15.4	17.2
Total	*143.8*	*71.2*	*73.8*	*78.6*
NO_x: 1000 tonnes				
Households	22.3	14.4	12.0	9.8
Energy	58.1	23.8	25.4	25.4
Services	8.1	7.6	7.4	7.0
Industry	61.2	42.5	42.2	42.2
Agriculture	10.3	12.0	9.5	7.5
Other	3.8	2.4	2.6	2.6
Refineries	17.7	8.1	8.2	8.2
Transport	313.1	161.2	149.3	173.5
Total	*494.6*	*272.1*	*256.6*	*276.2*
NH_3: 1000 tonnes				
Households	6.7	7.4	7.8	8.2
Industry	4.1	4.1	4.3	4.7
Agriculture	176.9	143.0	133.4	128.0
Other	0.6	0.7	0.8	0.8
Total	*188.4*	*155.2*	*146.3*	*141.7*

Source: NEO5 (draft report).

4.2.4 Monetary measure of impact

Due to the existence of an international treaty for transboundary air pollution, the relevant estimate is the damage to the 'world' due to emissions from the Netherlands.

It is also of interest to estimate the damage to the Netherlands from acidifiying pollutants released from the Netherlands and other countries and deposited in the Netherlands. In order to achieve this it is necessary to know the relationship between domestic emissions and deposition in the Netherlands, as well as foreign emissions and deposition in the Netherlands. Through impact measurement and valuation, it is then possible to estimate a damage per tonne deposited in the Netherlands. In order to achieve this, data have been combined from a variety of sources. Emissions data for the Netherlands for each of the acidifying pollutants have been provided by RIVM (*Table 4.2.2* above). This has been used in conjunction with the EMEP transfer matrices, MSC-W (1995), which allow us to estimate the proportion of emissions *from* the Netherlands which are deposited within and outside the country and therefore *damage caused by the Netherlands* within and outside the country. The relevant extracts from the EMEP data are provided in *Table 4.2.3* below.

Table 4.2.3 Geographic depositions of pollutants emitted from the Netherlands (1996)
(as a percentage of emissions from the Netherlands)

Pollutant	Receiving country	
	Netherlands	Elsewhere
NO_x (as N)	5%	95%
SO_2 (as S)	14%	86%
NH_3 (as N)	43%	57%

Source: http://projects.dnmi.no/-emep/

The reverse can also be estimated, i.e. the depositions to the Netherlands, caused by the Netherlands and by other countries. The data are provided in *Table 4.2.4* below.

Table 4.2.4 Sources of emissions of acidifying depositions to the Netherlands (1996)
(as a percentage of all depositions in the Netherlands)

Pollutant	Source of emissions	
	Netherlands	Elsewhere
NO_x (as N)	17%	83%
SO_2 (as S)	15%	85%
NH_3 (as N)	75%	25%

Source: http://projects.dnmi.no/-emep/

Unit damage values for SO_2 and NO_x and NH_3

Unit damage values for the acidifying pollutants have been based on the most recent ExternE report (1997) for the Netherlands, with some adjustments. The ExternE report estimates damages from pollutants emitted from the Netherlands for different fuel cycles. This produces a range of unit damage estimates, depending on the location of emissions and therefore depositions. We have used the results of the ExternE dispersion modelling exercise, but have made the following adjustments to the value estimates:

1. Notably, our estimates for mortality effects are based on acute mortality only. Chronic mortality effects are excluded due to the high scientific uncertainty surrounding these estimates, which are based on a single study (Pope et al, 1995) and have been interpreted in several different ways. Moreover, it is not clear how a reduction in *life expectancy* should be valued. The 'value of statistical life' (VOSL) approach is based on valuations of *mortality risk reduction*, as is the 'value of life year' (VOLY) approach which is derived from this. There is, in fact, only a single study which attempts to value changes in life expectancy (Johannesson and Johansson, 1996). The resulting estimate of willingness to pay for an increase in life expectancy of one year is approximately $600-1,500. However, extrapolating results from a single study in a single country (Sweden) is highly uncertain. Further original valuation research will undoubtedly be needed as the scientific estimates of chronic mortality effects improve. In our professional opinion, the current scientific and economic research in this area is too uncertain to be recommended for valuation at this time. These effects are therefore excluded from the current analysis.

2. Valuation of acute mortality effects for the over 65 age group is based on an age-adjusted VOSL (see *Annex 3 on Valuing Risk of Premature Mortality* for further details).

3. The unit values for a range of morbidity impacts have been taken from CSERGE et al (1999). These are more up-to-date estimates, and are based on a valuation study of the Netherlands specifically, whereas the values used in ExternE are adjusted cost of illness values taken from the USA. The CSERGE et al values are therefore likely to be much more accurate. These values are summarised in *Annex 4 on Monetary Valuation of Health Effects*.

The unit values resulting from these adjustments are summarised in *Table 4.2.5* below.

Table 4.2.5 Unit values for acidifying pollutants (Euro/tonne)

Pollutant	Effect	Unit Value (Euro/tonne) Lower	upper
NO_x	Mortality (via nitrates)	1,374	1,446
	Morbidity	764	840
	Total	*2,138*	*2,285*
SO_2	Mortality (via sulphates)	1,420	1,724
	Mortality (direct)	6,602	7,727
	Morbidity	753	955
	Crops	64	17
	Materials	0	365
	Monuments	311	6
	Total	*9,150*	*10,793*
NH_3	Mortality (via nitrates)	934	983
	Morbidity	520	571
	Total	1,454	1,554

Note that in the unit values for crops and monuments tend to move in the opposite direction from the health impacts. This is to be expected due to the different geographical locations of the receptors.

4.2.5 Aggregate monetary damage estimate

Applying the unit damage values summarised in *Table 4.2.5* to the emissions data given in *Table 4.2.2* results in estimates of total damages as shown in *Table 4.2.6* below.

Table 4.2.6 Damages to the UNECE including the Netherlands caused by the Netherlands acidifying pollutants: Million Euro/year

	1995	*2010*	*2020*	*2030*
Pollutant	*lower upper*	*lower upper*	*lower upper*	*lower upper*
NO_x	1,057 -1,130	582 -622	549 -586	591 -631
SO_2	1,316 -1,552	651 -768	675 -797	719 848
NH_3	274 -293	226 -241	213 -227	206 -220
Total	**2,647 -2,975**	**1,459 -1,631**	**1,437 -1,610**	**1,516 -1,700**

These estimates of damage are based on emissions from the Netherlands, and therefore should be interpreted as the damage caused by the Netherlands to the UNECE. This is evidently of relevance to policy-makers, since the Netherlands is signatory to the Convention on Long Range Transboundary Air Pollution. These are the relevant values for this study and for future cost benefit analysis studies.

For a measure of the importance of environmental issues as perceived by the population of the Netherlands, it is evidently deposition (and source) of pollution within the country that is of interest in the sense that this influences the Netherland's international negotiating strategy since the Netherlands cannot reduce these depositions itself. This may be calculated using the EMEP transfer matrices (*Tables 4.2.3 and 4.2.4*) in conjunction with the emissions data (*Table 4.2.2*) and adjusted unit values (*Table 4.2.5*). The results are presented in *Table 4.2.7*.

Table 4.2.7 Damages to the Netherlands due to acidifying pollutants from Holland and elsewhere. Million Euro/year

	1995		*2010*		*2020*		*2030*	
Pollutant	*lower*	*upper*	*lower*	*upper*	*lower*	*upper*	*lower*	*upper*
NO_x	311	332	171	183	161	172	174	186
SO_2	1,228	1,449	608	717	630	744	671	791
NH_3	157	168	130	138	122	130	118	126
Total	**1,696**	**1,949**	**908**	**1,038**	**914**	**1,046**	**963**	**1,103**

The results presented in *Table 4.2.7* are then combined with the information given in *Table 4.2.4* in order to establish; i) the damages to the Netherlands caused by emissions from the Netherlands and ii) the damage to the Netherlands from elsewhere. *Tables 4.2.8* and *4.2.9* present the results.

Table 4.2.8 Damages to the Netherlands due to acidifying pollutants from Holland: mEuro/year

	1995		*2010*		*2020*		*2030*	
Pollutant	*lower*	*upper*	*lower*	*upper*	*lower*	*upper*	*lower*	*upper*
NO_x	53	56	29	31	27	29	29	31
SO_2	184	217	91	108	95	112	101	119
NH_3	117	126	98	104	92	98	89	95
Total	**353**	**399**	**218**	**243**	**214**	**239**	**219**	**245**

Table 4.2.9 Damages to the Netherlands due to acidifying pollutants from elsewhere mEuro/year

	1995		*2010*		*2020*		*2030*	
Pollutant	*lower*	*upper*	*lower*	*upper*	*lower*	*upper*	*lower*	*upper*
NO_x	258	275	142	152	134	142	144	154
SO_2	1044	1232	517	609	536	632	570	672
NH_3	39	42	33	35	31	33	30	32
Total	**1341**	**1549**	**692**	**796**	**701**	**807**	**832**	**858**

Comparing the results of *Tables 4.2.6 , 4.2.8* and *4.2.9* demonstrate that for all of these pollutants, the Netherlands is a net exporter of pollution and therefore of damage. Additionally, due to the transport of emissions, cutting emissions of acidifying pollutants in the Netherlands will not be very successful in reducing national problems from acidification.

It should be noted that the analysis of acidification damage to the Netherlands assumes that the proportion of pollution coming from the Netherlands compared to other countries remains constant.

4.2.6 Uncertainty

The main sources of uncertainty are:

- the approach to premature mortality valuation;
- the approach to morbidity valuation, and
- omission of impacts on ecosystems, cultural assets (within materials damage) and visibility impacts. The impacts of NH_3 emissions relate only to health and agriculture, i.e. impacts to ecosystems through eutrophication are omitted. However, eutrophication in the Netherlands is assessed in *Section 4.6.*

In response to the first two areas of uncertainty we conduct a sensitivity analysis, see below. The omission of impacts with potentially large benefits from the control of acidifying pollutants suggests that the overall direction of error in the benefit estimates is biased downwards. However, benefits to health tend to dominate the results. It is also important to note that the damage assessment for acidification may be overestimated due to the inclusion of health related PM10 impacts which are already accounted for in the separate analysis on PM10.

The sensitivities considered are as follows:

1. *Effect of assuming different values for a VOSL*. The valuation of mortality effects is conducted using a 'value of statistical life'. For more details on this approach see *Annex 3 on Valuing Risk of Premature Mortality*. Mortality incidence in the over 65 age group is valued at 70% of a VOSL, based on the results of Pearce (1997). The sensitivity explores three different assumptions about the value of a VOSL, namely:

- the effect of removing the age adjustment;
- using the latest VOSL estimate by CSERGE et al (1999) with age adjustment, and
- using the CSERGE estimate without age adjustment.

2. *Effects of adopting older unit value estimates for valuation of morbidity effects*: The baseline results are based on the latest unit damage estimates from CSERGE et al., (1999). These are the most up-to-date estimates of WTP to avoid these impacts, and are also specific to the Netherlands. For these reasons, the damage estimates produced are likely to be much more accurate than previous estimates. However, most cost-benefit assessments to date have been based on a different set of unit values, adjusted from American studies. These are summarised in *Annex 4 on Monetary Value of Health Effects*. Therefore, for purposes of comparison of our results with previous studies, we explore the effect of using the older set of estimates.

The effects of changing each of these assumptions on the unit values for different pollutants are summarised in *Table 4.2.10* below. The overall minimum and maximum unit damages reported for each pollutant are derived by using the different assumptions, as outlined above, which when combined produce the highest and the lowest estimates.

Table 4.2.10 Unit values for sensitivity analysis (Euro/tonne)[11]

Pollutant	Effect	Unit Value (Euro/tonne)	
		lower	upper
NO_x	Mortality (baseline)	1,374	1,446
	no age adjustment	1,952	2,054
	CSERGE age adjusted	1,961	2,064
	CSERGE no age adjustment	2,802	2,949
	Morbidity (baseline)	764	840
	Morbidity (ExternE)	687	755
Overall min and max, combining assumptions		*2,061*	*3,789*
SO_2	Mortality (via sulphates)	1,420	1,724
	no age adjustment	2,016	2,448
	CSERGE age adjusted	2,029	2,463
	CSERGE no age adjustment	2,894	3,514
	Mortality (direct, baseline)	6,602	7,727
	no age adjustment	9,378	10,976
	CSERGE age adjusted	9,432	11,039
	CSERGE no age adjustment	13,459	15,752
	Morbidity (baseline)	753	955
	ExternE	676	857
	Crops	64	17
	Materials	0	365
	Monuments	311	6
Overall min and max, combining assumptions		*9,073*	*20,609*
NH_3	Mortality (via nitrates)	934	983
	no age adjustment	1,327	1,397
	CSERGE age adjusted	1,333	1,403
	CSERGE no age adjustment	1,905	2,005
	Morbidity (baseline)	520	571
	Morbidity (ExternE)	467	514
Overall min and max, combining assumptions		*1,401*	*2,576*

The sensitivity analysis for the total damage to the UNECE area due to acidifying emissions from the Netherlands has been produced by combining the above assumptions to provide overall lower bound and upper bound unit values. These are combined with the emissions data to produce the total damage estimates presented in *Table 4.2.11* below.

Table 4.2.11 Damages to the UNECE including the Netherlands due to acidifying pollutants from the Netherlands (mEuro/year)

	1995	***2010***	***2020***	***2030***
Pollutant	*lower upper*	*Lower upper*	*lower upper*	*lower upper*
NO_x	1,019 -1,874	561 -1,031	529 -972	569 -1,047
SO_2	1,305 -2,964	646 -1,467	670 -1,521	713 -1,620
NH_3	264 -485	217 -400	205 -377	199 -365
Total	2,588 -5,323	1,424 -2,898	1,403 -2,870	1,481 -3,031

[11] Note that in the sensitivity analysis the unit values for crops and monuments tend to move in the opposite direction from the health impacts. This is to be expected due to the different geographical locations of the receptors. However, the upper and lower bounds are therefore not based on the addition of upper and lower bound unit values for individual receptor, but the maximum and minimum possible combinations of values these may take.

Appendix Assumptions behind the unit damage values

In the report for the Netherlands in the ExternE study (1997) IVM estimates damages to the UNECE area from emissions from the Netherlands by following the ‘bottom-up’ methodology described in *Section 3*. This means many variables are estimated and assumptions are made which ultimately remain embedded in the final unit damage values used in this study. For the purposes of transparency we present, where possible, the variables used and the assumptions made in their analysis.

Stock at risk

ExternE (1998) estimate the fraction of the Europeans and the UK population in the various groups considered to be at special risk such as the elderly, children and asthmatics, i.e. the variable 'STOCK' mentioned in *Section 3*.

Fraction of children in European population:	0.2
Fraction of adults in European population:	0.8
Fraction of people > 65 years in European population:	0.14
Annual death rate per thousand people in Europe:	10.2
Child asthmatics as fraction of UK population:	0.02
Adult asthmatics as fraction of UK population:	0.04

These fractions are used in estimating the ‘STOCK’ variable (the relevant fraction multiplied with the total population) if the 'bottom-up' method is adopted.

Exposure response functions

The UNECE unit damage values used in this study contain within them damage to human health, materials and agriculture. The quantification of human health impacts, damage to materials and the effects of air pollution on agricultural symptoms are estimated using the exposure response relationships given below.

Human health: In *Table 4.2.12*, the coefficient ‘b’ is interpreted as the increase in annual incidence of each symptom. For example, i) for morbidity: the coefficient is the number of cases / year.person.$\mu g/m^3$, ii) for acute mortality the coefficient b is the % change in mortality / rate. $\mu g/m^3$ and iii) for chronic mortality, b is the years of life lost for chronic effects on mortality.

Table 4.2.12 Exposure response coefficients for health impacts

Receptor	Impact category	Reference	Pollutant	b
ASTHMATICS				
Adults	Bronchodilator usage	Dusseldorp et al., 1995	PM10	0.163
	Cough	Dusseldorp et al., 1995	PM10	0.168
	Lower respiratory symptoms (wheeze)	Dusseldorp et al., 1995	PM10	0.061
Children	Bronchodilator usage	Roemer et al, 1993	PM10	0.078
	Cough	Pope, Dockery, 1992	PM10	0.133
	Lower respiratory symptoms (wheeze)	Roemer et al., 1993	PM10	0.103
All	Asthma attack	Whittemore and Korn, 1980	O_3	4.29×10^{-3}
ELDERLY 65 YEARS +				
	Congestive heart failure	Schwartz and Morris, 1995	PM10	1.85×10^{-5}
CHILDREN				
	Chronic bronchitis	Dockery et a.l, 1989	PM10	1.61×10^{-3}
	Chronic cough	Dockery et al., 1989	PM10	2.07×10^{-3}
ADULTS				
	Restricted activity days	Ostro, 1987	PM10	0.025
	Minor restricted activity days	Ostro and Rothschild, 1989	O_3	9.76×10^{-3}
	Chronic bronchitis	Abbey et al., 1995	PM10	4.9×10^{-5}
ENTIRE POPULATION				
	Respiratory hospital admission	Dab et al., 1996 Ponce de Leon, 1996	PM10 SO_2 O_3	2.07×10^{-6} 2.04×10^{-6} 7.09×10^{-6}
	Cerebrovascular hospital admissions	Wordley et al., 1997	PM10	5.04×10^{-6}
	Symptom days	Krupnick et al., 1990	O_3	0.033
DEATH RATES				
	Acute mortality	WHO, 1997	PM10	0.074%
	Acute mortality	Anderson et al., 1996 Touloumi et al., 1996 Sunyer et al., 1996	SO_2 O_3	0.072% 0.059%
	Chronic mortality	Pope et al., 1995	PM10	0.00036

Source: ExternE, European Commission, 1995b, 1998)

Materials: AEA Technology (1999) report that the dose response functions used are derived mainly from the UNECE Programme (Kucera, 1993a, 1993b, 1994), unless otherwise referenced. *Table 4.2.13* lists the functions.

The following key applies to all the relationships given:

ER	Erosion rate (um / year)
P	Precipitation rate (m/year)
SO_2	Sulphur dioxide concentration ($\mu g/m^3$)
O_3	Ozone concentration ($\mu g/m^3$)
H^+	Acidity (meq/m^2/year
R_H	Average relative humidity, (%)
F_1	$1-\exp(-0.121.R_H/(100-R_H))$
TOW	Fraction of time relative humidity exceeds 80% and temperature >0°C
ML	Mass loss(g/m^2) after 4 years

In all the relationships, the original H^+ concentration term (in mg/l) is replaced by an acidity term, using the conversion: $P.H^+$ (mg/l) = $0.001.H^+$ (acidity in meq/m^2/year). To convert mass loss for stone and zinc into an erosion rate in terms of material thickness, respective densities of 2.0 and 7.14 tonnes / m^3 are assumed. The relationships are given in *table 4.2.13*.

Table 4.2.13 Exposure response functions for materials damage

Unsheltered limestone (4 years)	$ML = 8.6 + 1.49.TOW.SO_2 + 0.097.H^+$
Unsheltered sandstone (4 years) (also mortar)	$ML = 7.3 + 1.56.TOW.SO_4 + 0.12.H^+$
Brickwork	No effect
Concrete	Assumed no effect, though air pollution may affect steel reinforcement
Carbonate paint, (Haynie, 1986)	$\Delta ER/tc = 0.01(P)8.7(10^{-pH} - 10^{-5.2})+0.006.SO_2.f_1$
Silicate paint, (Haynie, 1986)	$\Delta ER/tc =0.01(P)1.35(10^{-pH} - 10^{-5.2})+0.00097.SO_2.f_1$
Steel	Assumed either pained or galvanised, not assessed independently
Unsheltered zinc (4 years)	$ML = 14.5 + 0.043.TOW.SO_2.O_3+ 0.08.H^+$
Sheltered zinc (4 years)	$ML = 5.5 + 0.013.TOW.SO_2.O_3$
Aluminium	Assumed too corrosion resistant to be affected significantly

Agriculture: crops and pasture grass: The ExternE unit damage values include the four major impacts to agricultural systems, i) acidifying soils / liming, ii) N deposition as fertiliser, iii) direct effects of SO_2 and O_3 on crop yield and iv) indirect SO_2 and O_3 effects on livestock. Quantification of the first two impacts follows a simple methodology, the former measures the additional costs of liming at Euro 16.8 per tonne of lime, and the latter measures the cost savings of reduced nitrogen fertiliser at Euro 430 per tonne nitrogen (Nix, 1990). For further details refer to AEA Technology (1999).

The following functions are used to quantify the % yield change from SO_2 effects on different crops[12]. These functions take into account the fertilisation effect of sulphur at low concentrations.

From 0 to 13.6 ppb SO_2: $\Delta = 0.74(SO_2) - 0.55(SO_2)^2$

Above 13.6 ppb SO_2: $\Delta = -0.69(SO_2) +9.35$

[12] Such as: maize, oats, leaf crops, soybeans, sunflower, barley, wheat, rice, millet, potato, linseed, tomato, hops, tobacco, rye, sugar beet, beans, carrots, hemp, raspberries, cucumber, sorghum, strawberries, flax, sesame seeds.

Whilst for pasture the following exposure response functions are assumed:

From 0 to 15.3 ppb: $\Delta = 0.20(SO_2) - 0.013(SO_2)^2$
Above 15.3 ppb: $\Delta = -0.18(SO_2) + 2.75$

Agriculture: livestock: the impacts of acidifying pollutants to meat and milk production are assumed to be 50% as sensitive to pasture grass (AEA Technology, 1999).

4.3 Noise

4.3.1 The issue

In urban areas, noise pollution is one of the main local environmental issues. The full health effects of noise have not been fully explored, but are known to have ‘annoyance’ effects on humans. This ‘annoyance’ can have effects on the quality of sleep, communication as well as psycho-physiological behaviour, including stress and at higher levels cardiovascular reactions. At very high sound levels this annoyance is extended into definite physiological effects such as, hearing loss. Other possible health effects due to exposure to noise include blood pressure changes, hormone level fluctuations and effects on the immune system. In OECD countries, the issue of unacceptable noise levels is ranked as a severe or very severe problem by 25% of cities and is due to get worse as the urban population increases. Noise also has effects on wildlife. These have not been fully documented but research points to extra auditory effects, mainly unspecified stress reactions, (mainly on animals with an acute sense of hearing) under extremely high noise levels from low-flying aircraft.

4.3.2 Source of emissions

The major source of noise is road, air and rail traffic. Of these, road traffic accounts for the highest percentage of household exposure with the main source of noise being tyre / road noise. Of road traffic, city roads have the greatest percentage of household exposure followed by high-ways and then regional roads. Road traffic is due to worsen as the increased density of road traffic more than makes up for any progress in tightening limits and progression in technology to reduce noise. Other sources of noise pollution include industry and recreation, as well noise emissions from construction sites and from traffic on waterways. Although these sources are less constant, they remain significant in certain areas.

4.3.3 Physical measure of impacts

Population exposure to noise nuisance by dB(A) bands

Data on percent population exposure to road traffic, airport and railway noise are presented in *Table 4.3.1*[13].We convert the data to numbers of households exposed to noise by dividing the number of millions of people exposed to the various noise bands, i.e. by using population data for 1995, 2000, 2010, 2030,[14] by the estimated household size (in 1995, 2010, 2020, 2030)[15]. *Table 4.3.2* provides the adjusted data.

[13] This study assumes that noise levels below 50dB are not considered a nuisance, however, the authors recognise that there are some studies that indicate the threshold for noise levels should be nearer 40dB.
[14] Population data: 1995- 15.4m, 2010- 16.8m, 2020- 17.7m, 2030-18.4m, source RIVM (2000).
[15] Household size: 1995- 2.34, 2010- 2.27, 2020- 2.23, 2030- 2.21, source RIVM (2000).

Table 4.3.1 Percent of Dutch population exposed to various noise bands from road, rail and aircraft

dB(A)	51-55	56-60	61-65	66-70	71-75	76-80	>80	% exposed >50dB(A)
1995								
TOTAL	29.17	27.03	11.68	2.76	0.47	0.10	0.01	71.2
Road transport total	28.45	21.96	8.70	1.71	0.17	0.02	0.00	61.0
- highways	11.12	4.27	1.20	0.34	0.11	0.02	0.00	17.1
- regional roads	7.07	3.45	1.74	0.69	0.03	0.00	0.00	13.0
- city roads	16.87	12.76	4.98	0.55	0.00	0.00	0.00	35.2
Rail	8.06	3.85	1.80	0.77	0.26	0.08	0.01	14.8
Air	4.72	1.38	0.17	0.03	0.00	0.00	0.00	6.3
2010								
TOTAL	28.19	29.65	14.22	3.84	0.74	0.15	0.02	76.8
Road transport total	28.85	23.37	10.17	2.41	0.30	0.02	0.00	65.7
- highways	12.27	4.44	1.23	0.39	0.11	0.02	0.00	18.5
- regional roads	8.67	4.28	2.09	1.04	0.14	0.00	0.00	16.2
- city roads	17.40	12.95	5.94	0.86	0.02	0.00	0.00	37.2
Rail	10.11	4.78	2.13	0.98	0.37	0.12	0.02	18.5
Air	8.70	3.19	0.36	0.05	0.01	0.00	0.00	12.3
2020								
TOTAL	26.66	30.44	15.15	4.36	0.91	0.16	0.04	77.7
Road transport total	28.15	24.33	10.90	2.89	0.47	0.03	0.00	66.8
- highways	13.34	5.44	1.62	0.48	0.16	0.03	0.00	21.1
- regional roads	9.60	4.84	2.40	1.21	0.24	0.00	0.00	18.3
- city roads	17.00	12.55	5.86	1.05	0.03	0.00	0.00	36.5
Rail	9.61	4.84	2.11	0.94	0.36	0.11	0.03	18.0
Air	9.24	3.81	0.49	0.11	0.01	0.00	0.00	13.7
2030								
TOTAL	25.86	31.42	16.25	4.88	1.13	0.19	0.04	79.8
Road transport total	27.87	25.90	11.83	3.45	0.68	0.05	0.00	7.0
- highways	14.50	6.58	1.95	0.59	0.19	0.04	0.00	23.9
- regional roads	10.45	5.31	2.65	1.36	0.35	0.01	0.00	20.1
- city roads	17.42	12.83	6.03	1.33	0.08	0.00	0.00	37.7
Rail	9.16	4.65	2.09	0.88	0.35	0.12	0.03	17.3
Air	9.74	4.16	0.60	0.12	0.01	0.00	0.00	14.6

Source: NEO5, (draft report).
Note, *Table 4.3.1* shows the percent population exposure data to different noise bands above 50dB(A) only. This study assumes that exposure to noise levels below 50dB(A) is not considered a nuisance.

Table 4.3.2 confirms that the main source of noise is from roads. Over time, percent of population exposed to the lower noise bands, i.e. 0 - 40dB(A), generally falls. However, we see that between 1995 and 2030, percentage of population exposure to the higher noise bands increases. For example, there is an increasing percentage of population exposure to total noise from 56dB(A) onwards, likewise for roads. Whilst, there is an increasing percentage of population exposure to rail noise from 51dB(A) onwards and for air, from 41dB(A) onwards.

Table 4.3.2 Dutch households exposed to road, rail and aircraft noise: thousand households

DB(A)	**51-55**	**56-60**	**61-65**	**66-70**	**71-75**	**76-80**	**>80**
1995							
Total	1923	1782	770	182	31	7	1
Road transport total	1875	1447	573	113	11	2	0
- high ways	733	281	79	23	7	1	0
-regional roads	466	227	115	46	2	0	0
-city roads	1112	841	328	36	0	0	0
Rail	531	254	119	51	17	5	1
Airport	311	91	11	2	0	0	0
2010							
Total	2089	2197	1054	285	55	11	1
Road transport total	2138	1732	91	179	22	1	0
- high ways	910	329	155	29	8	2	0
-regional roads	642	317	440	77	10	0	0
-city roads	1289	960	158	64	1	0	0
Rail	749	355	27	73	27	9	2
Airport	645	236	0	4	1	0	0
2020							
Total	2113	2412	1201	346	72	13	3
Road transport total	2231	1928	864	229	37	2	0
- high ways	1057	431	128	38	13	2	0
-regional roads	761	384	190	96	19	0	0
-city roads	1347	995	464	83	2	0	0
Rail	762	384	167	74	29	9	2
Airport	732	302	39	9	1	0	0
2030							
Total	2154	2617	1353	406	94	16	3
Road transport total	2321	2157	985	287	57	4	0
- high ways	1208	548	163	49	16	3	0
-regional roads	870	442	221	113	29	1	0
-city roads	1451	1069	502	111	7	0	0
Rail	763	388	174	73	30	10	3
Airport	811	346	50	10	1	0	0

Between 1995 and 2030, household exposure to noise bands between 51d(B) and 80d(B) increases for all forms of noise.

4.3.4 Monetary measure of impacts

A large weight of evidence now exists to support the contention that noise pollution is capitalised in the price of property. There are two appropriate methodologies for estimating the size of this impact: the hedonic property price approach and contingent valuation. While other techniques have been used, these two are the main ones.

The hedonic property price methodology for evaluating the aggregate monetary value cost of noise nuisance is as follows:

Noise cost per annum = ((POP/HHOLDSIZE).HPRICE.NSDI (dB(A) - 50)) / a

Where;

POP	=	population exposed to noise in a given noise band;
HHOLDSIZE	=	household size in persons/household;
HPRICE	=	average house price;
NSDI	=	noise sensitivity depreciation index, i.e. percentage depreciation in house price for each decibel (dB(A)) of noise above the baseline level;
dB(A)	=	actual decibel exposure level;
50	=	50dB(A), the assumed baseline noise exposure level below which NSDI = 0, i.e. noise is not considered a nuisance below 50dB(A);
a	=	annuitisation factor, i.e. a factor converting house prices which are present values, into annual values.

Apart from population exposure data and the choice of the noise threshold level (e.g. 50 dB(A)(A)) the critical parameter in the equation is the NSDI.

European noise valuation studies

Tables 4.3.3, *4.3.4* and *4.3.5* show recent studies on noise nuisance for road traffic noise, airport noise and railway disturbance. Only post 1990 studies are included and only European studies are shown. Some tentative conclusions can be derived from these.

First, a simple average of the road traffic studies suggests a NSDI of 0.6-0.8 and this in fact corresponds to the values that are frequently ‘borrowed’ for benefits transfer estimates. However, recent very thorough studies such as, Bateman et al (1999) for the UK and Vainio (1995) for Finland point to much lower estimates in the range 0.2-0.4 for the NSDI.

Second, the few estimates derived from contingent valuation studies are remarkably consistent with estimates derived from ‘aggregate studies’ (see later) which suggest that figures like Euro15 - 20 per person, per annum, per ‘excess’ decibel is a consensus value.

Third, there is some suggestion in the literature that values for noise nuisance from aircraft may be higher than for road noise nuisance. *Table 4.3.4* certainly contains some very high values for the NSDI.

Table 4.3.3 Road traffic noise valuation studies in Europe

Study	Country	NSDI hedonic % of house price per dB(A)	NSDI cvm i.e. WTP (Euro 2000)
Vainio (1995) Haolomo (1992)	Finland	0.36 0.98	0.7-1.05
Weinberger et al. (1991)	Germany	0.5 - 1.3	13 pp pa pdB(A)
Collins and Evans (1994) Bateman et al. (1999)	UK UK	0.65-1.28 0.20	-
Soguel (1994) Pommerehne (1988) Iten and Maggi (1990)	Switzerland	0.91 1.26 0.9	16 pp pa pdB(A) -
Saelensminde and Hammer (1994) Grue et al. (1997) Obos Flats Houses	Norway	0.24-0.54 - 0.24 0.21 0.54	30-54 pp pa pdB(A)
Lambert (1992)	France	1.0	-

Table 4.3.4 Aircraft noise valuation studies in Europe

Study	Country	NSDI hedonic. % of property price per dB(A)	NSDI cvm Euro (2000)
Pennington et al., 1990 Tomkins et al., 1995 Collins and Evans, 1994 Yamaguchi, 1996 Heathrow Gatwick Bateman et al 1999	UK	0 0.47 0 1.51 2.30 0.20	
Feitelson et al., 1995 *Owners* *Renters*	Israel	 2.4- 4.1 1.8-3.0	

Table 4.3.5 Railway noise valuation studies in Europe

Study	Country	NSDI hedonic. % of property price per dB(A)	NSDI cvm Euro (2000)
Strand, 1999 *Hedonic study* *Estate valuers*	Norway	 0.1 * -	 - 289 per metre from line

* note: this is the percentage change in property price per metre change in distance from the railway line.

Dutch noise valuation studies

Verhoef (1996) notes that there is limited research on traffic externalities in the Netherlands. Bonenschansker et al (1995) and Bleijenberg et al. (1994) appear to be the two main recent studies. The former is an attempt to estimate damage costs while the latter assembles existing studies and applies them to the Dutch context. The estimates are summarised in *Table 4.3.6*:

Table 4.3.6 Road and rail noise valuation studies in the Netherlands: Euro million (2000)

Study	Road noise damage	Rail noise damage
Bonenschansker et al (1995)	260 - 572	5
Bleijenberg et al (1994)	334 (104 - 569)	10 (5 - 16)

Note: lower / upper values given in brackets.

Verhoef (1996) judges the Bonenschansker et al. (1995) estimates for noise to be better than those of Bleijenberg et al. (1994), since they make use of house price depreciation estimates. He also judges the higher end of the scale to be more likely. Taking the Euro million 572 estimate would therefore suggest a per capita valuation of some Euro 37 (assuming a population of 15.4 million)**.** This is well in accord with the consensus per capita estimates to be discussed later in 'aggregate studies'.

Meta studies

Bertrand (1997) conducts a meta-analysis[16] of noise valuation studies and finds a fitted equation of:

$$\text{MWTP} = e^{2.348+0.00000509Y + 0.0497n}$$

where *MWTP* is marginal willingness to pay, *Y* is income and *n* is noise in dB(A). Given the mean values for *Y* and *n* (Euro = 20,300 in 1995 (2000 prices) and 71.8 dB(A)) this gives an income elasticity of MWTP of 0.1 and an elasticity of MWTP with respect to noise of 3.57. In other words, WTP increases by 0.1% for each 1% increase in income, and WTP increases by just under 4% for each dB(A) increase in noise. The link between WTP and noise is thus non-linear, confirming what a number of authors have suggested (e.g. Christensen et al, 1998) and in contrast with the assumption in ECMT (1998).

Thus, to use the Betrand's meta-equation, the procedure would be, for example:

Taking the number of households exposed to 50-55dB(A), of which there are 1.923 million in 1995. A single household's WTP to go from an average of 53dB(A) to 50dB(A) is:

$e^{2.451 + 2.6341} - e^{2.451+2.485}$ = Euro 22.35 per household x 1.923 million households = Euro 43 million for this group alone. It is necessary to repeat this for each group in each noise band in 1995 and then add up the total for 1995 damage estimates. For example, for noise band 56 -60 the calculation would be done for N=58 down to N=50, i.e. we assume we require their WTP for a reduction to background levels of noise, and so on. For later years, it is necessary to use the population exposure data for 2010, 2020 and 2030 to the different noise bands, and change the income level to allow for percentage growth in GDP.

Bertrand (1997) demonstrates that the price for noise will be higher in property markets where households are relatively more wealthy and where the general level of noise pollution is relatively higher. *Table 4.3.7* presents the variation of WTP with different noise levels, based on Bertrand's meta-equation.

[16] Meta-analysis is a study of other studies.

Table 4.3.7 WTP variation with different noise levels

Noise band	average noise level	WTP per household Euro
51 – 55	53	22.4
56 – 60	58	67.9
61 – 65	63	126.4
66 – 70	68	201.4
71 – 75	73	297.4
76 – 80	78	420.6
>81	83	578.5

Source: Authors.

Schipper (1998) conducts a meta-analysis of 30 aircraft noise hedonic house price studies. His resulting equation is:

NSDI= -1.54 +0.3 (H/Y) –0.4 (if log-linear form) + 0.01

Where *H* is the average house price and *Y* is average income.

Aggregate studies

Maddison et al. (1996) suggest that road traffic noise costs in the UK are about £2.6 billion p.a., some Euro 4.0 billion. This study assumes an NSDI of 0.0067 and an average house price of Euro 73,000 is used. Taking the numbers of people exposed to noise in each noise interval suggests that road traffic contributes some 191 million 'excess' decibels. Hence the WTP to avoid each decibel is 4.0 billion/191 million = Euro 21 per person per year per decibel. This matches the guideline figure suggested by The Task Force for the European Conference of Ministers of Transport (ECMT, 1998) of Euro 21 per person per annum per dB(A) above the threshold. While the Maddison et al. figure is for road transport only and the ECMT figure is for road, rail and air; values per unit of noise could be assumed to be similar, although there is some suggestion that aircraft and rail noise are regarded as being more annoying than road noise (e.g. Strand, 1999).

Johansson (1996) reports estimates for Sweden of some 2.6 billion SEK per year for road traffic only, or some Euro 325 million per annum. However, Johansson notes that the Swedish estimates are based on an early (1970s) hedonic price study and may be unreliable. If the total figure is accepted, WTP would be some Euro 37 pp/pa/pdB(A).

Verhoef (1996) reports Dutch estimates of some NLG 660 million for road and rail noise, or some Euro 329 million, Euro 21 pp/pa/pdB(A) (assuming a population of 15.4 million).

The strong similarity between the estimates is slightly illusory since the per person WTP estimates are derived from reviews of the available valuation studies. Nonetheless, there is a remarkable consistency about them, suggesting that a default figure of around Euro15 – 20 pp/pa/pdB(A) above threshold could be used for valuation purposes. Similarly, Bertrand's meta-analysis suggests a value that rises with the noise level. At around 60 dB(A) WTP would be Euro 32 and at 70 dB(A) the WTP would be closer to Euro 53.

4.3.5 Aggregate monetary damage estimate

Total noise damage

Two approaches are used to estimate the noise damage value in the Netherlands. The first is the unit damage value approach and uses the WTP per 'excess' decibel values derived from contingent valuation studies. The second approach makes use of the hedonic property price approach. The results are given below.

Unit damage value approach
The estimates of the economic value of noise nuisance presented earlier are sufficiently close that ‘default’ values of a willingness to pay per person per annum per ‘excess’ decibel can be used in ‘benefits transfer’ contexts. Based on the literature review presented above, we suggest that the default value is around Euro 20 pp/pa/pdB(A). Table 4.3.8 reports the total noise nuisance damage in the Netherlands, based on this figure.

In 1995, noise nuisance is estimated to be Euro billion 1.6, in 2010 Euro billion 1.8, in 2020 Euro billion 1.9 and in 2030 Euro billion 20. Noise damage is shown to be increasing between the years 1995 and 2030 because there is an increasing population exposed to noise nuisance for all noise bands between 56 and 80 dB(A).

Table 4.3.8 Total noise damage for the Netherlands in 1995, 2010, 2020 and 2030: Euro million

Noise band dB(A)	Average exceedance		Population exposed Million		WTP Euro		Noise damage Euro million
1995							
51-55	3	X	4.50	X	20	=	270
56-60	8	X	4.17	X	20	=	667
61-65	13	X	1.80	X	20	=	468
66-70	18	X	0.43	X	20	=	153
71-75	23	X	0.07	X	20	=	33
76-80	28	X	0.02	X	20	=	9
>80	32.5	X	0.00	X	20	=	1
Total							***1602***
2010							
51-55	3	X	4.35	X	20	=	261
56-60	8	X	4.57	X	20	=	732
61-65	13	X	2.19	X	20	=	570
66-70	18	X	0.59	X	20	=	213
71-75	23	X	0.11	X	20	=	53
76-80	28	X	0.02	X	20	=	13
>80	32.5	X	0.00	X	20	=	2
Total							***1844***
2020							
51-55	3	X	4.11	X	20	=	247
56-60	8	X	4.70	X	20	=	751
61-65	13	X	2.34	X	20	=	608
66-70	18	X	0.67	X	20	=	242
71-75	23	X	0.14	X	20	=	65
76-80	28	X	0.02	X	20	=	14
>80	32.5	X	0.01	X	20	=	4
Total							***1930***
2030							
51-55	3	X	3.99	X	20	=	239
56-60	8	X	4.85	X	20	=	775
61-65	13	X	2.51	X	20	=	652
66-70	18	X	0.75	X	20	=	271
71-75	23	X	0.17	X	20	=	80
76-80	28	X	0.03	X	20	=	16
>80	32.5	X	0.01	X	20	=	4
Total							***2038***

Hedonic property price approach
To calculate the capital value of noise damage for the Netherlands following the hedonic property price approach, we multiply the number of households exposed to average noise exceedance above 50dB(A) in each noise band, by the average house price in the Netherlands and by the NSDI value. We illustrate the methodology by considering *total* noise (e.g. rail, road and aircraft) in the Netherlands between 1995 and 2030.

Table 4.3.9 demonstrates the total noise damage calculation for the Netherlands. For the main analysis we assume:

- percentage population exposed to different noise bands (see *table 4.3.1*);
- population in the Netherlands in 1995, 2010, 2020 and 2030 assumed to be 15.4m, 16.8m, 17.7m and 18.4m respectively, (see *Annex 5 on Environmental Data, Assumptions and Models*);
- size of households in 1995, 2010, 2020 and 2030 is 2.34, 2.27, 2.23 and 2.21 respectively (see *Annex 5 on Environmental Data, Assumptions and Models*);
- average house price is Euro 124,921;
- NSDI is assumed to be 0.4, and
- threshold below which noise levels are not considered a nuisance is 50dB(A);

The influence of each parameter on the final result is tested in the sensitivity analysis in *Section 4.3.5*.

Table 4.3.9 Total noise damage for the Netherlands in 1995, 2010, 2020 and 2030

1995					
Noise band	Average exceedance	No of households	NDSI	Average house price	Damage Present value
dB(A)	DB(A)	thousands	mid	Euro (2000)	Euro million (2000)
51-55	3	1923	0.004	124921	2882.3
56-60	8	1782	0.004	124921	7122.2
61-65	13	770	0.004	124921	5001.1
66-70	18	182	0.004	124921	1636.3
71-75	23	31	0.004	124921	356.0
76-80	28	7	0.004	124921	92.2
>80	32.5	1	0.004	124921	10.7
TOTAL					***17100.8***
2010					
51-55	3	2089	0.004	124921	3131.4
56-60	8	2197	0.004	124921	8782.9
61-65	13	1054	0.004	124921	6844.9
66-70	18	285	0.004	124921	2559.3
71-75	23	55	0.004	124921	630.2
76-80	28	11	0.004	124921	155.5
>80	32.5	1	0.004	124921	24.1
TOTAL					***22128.2***
2020					
51-55	3	2113	0.004	124921	3167.2
56-60	8	2412	0.004	124921	9643.5
61-65	13	1201	0.004	124921	7799.3
66-70	18	346	0.004	124921	3107.8
71-75	23	72	0.004	124921	828.8
76-80	28	13	0.004	124921	177.4
>80	32.5	3	0.004	124921	51.5
TOTAL					***24775.7***
2030					
51-55	3	2154	0.004	124921	3228.6
56-60	8	2617	0.004	124921	10460.7
61-65	13	1353	0.004	124921	8791.4
66-70	18	406	0.004	124921	3655.6
71-75	23	94	0.004	124921	1081.6
76-80	28	16	0.004	124921	221.4
>80	32.5	3	0.004	124921	54.1
TOTAL					***27493.3***

Note, the total damage values given in *Table 4.3.9* are based on non-rounded population figures, thus the results may differ if total damage is calculated based on information given in *Table 4.3.9* alone.

The total cost figures presented in *Table 4.3.9* are present values, these need to be annuitised to obtain an annual damage cost. We choose a 6% real annuity rate to reflect actual interest rates in the housing market and a lifetime of 35 years. The annuitisation factor is then 14.49. (See *Definitions*). The annual total noise cost for the Netherlands are then Euro billion 1.1 in 1995, Euro billion 1.5 in 2010, Euro billion 1.7 in 2020 and Euro billion 1.8 in 2030 (i.e. total noise cost divided by the annuitisation factor).

The noise damage estimates based on the hedonic property price approach are slightly lower than those estimates based on the unit damage values for noise. This may be explained by the fact that using the hedonic price approach values noise only in the context of the home. This suggests the hedonic property price results may be biased downwards.

In order to see the level of damage caused by the different sources of noise, the following sections report the noise damage estimates for road, rail and air separately based on the hedonic property price approach.

Road noise damage estimates

Table 4.3.10 presents the noise costs due to road traffic only according to the hedonic property price approach.

Table 4.3.10 Road noise damage costs in the Netherlands: Euro million (2000), present value

Noise band: dB(A)	1995	2010	2020	2030
All roads				
51-55	2811.1	3204.7	3344.3	3479.5
56-60	5786.3	6922.6	7707.8	8622.9
61-65	3725.1	4895.4	5611.4	6400.2
66-70	1013.8	1606.2	2060.0	2584.4
71-75	128.8	255.5	428.1	650.9
76-80	18.4	20.7	33.3	58.3
>80	0.0	0.0	0.0	0.0
Present value	*13483.5*	*16905.2*	*19184.9*	*21796.1*
Total annual cost	**930**	**1166**	**1323**	**1503**
Highways				
51-55	1098.8	1363.0	1584.8	1810.3
56-60	1125.1	1315.2	1723.4	2190.7
61-65	513.8	592.1	834.0	1055.0
66-70	201.6	259.9	342.1	442.0
71-75	83.3	93.7	145.7	181.9
76-80	18.4	20.7	33.3	46.6
>80	0.0	0.0	0.0	0.0
Present value	*3041*	*3645*	*4663*	*5726*
Sub-total annual cost	***210***	***251***	***322***	***395***
Regional roads				
51-55	698.6	963.1	1140.5	1304.7
56-60	909.0	1267.8	1533.3	1767.9
61-65	745.0	1006.0	1235.5	1433.7
66-70	409.1	693.2	862.5	1018.8
71-75	22.7	119.2	218.6	335.0
76-80	0.0	0.0	0.0	11.7
>80	0.0	0.0	0.0	0.0
Present value	*2784*	*4049*	*4991*	*5872*
Sub-total annual cost	***192***	***279***	***344***	***405***
City roads				
51-55	1666.9	1932.8	2019.6	2174.9
56-60	3362.2	3836.0	3975.9	4271.5
61-65	2132.3	2859.2	3016.8	3262.3
66-70	326.1	573.2	748.4	996.3
71-75	0.0	17.0	27.3	76.6
76-80	0.0	0.0	0.0	0.0
>80	0.0	0.0	0.0	0.0
Present value	*7487.4*	*9218.3*	*9788.1*	*10782*
Sub-total annual cost	***516***	***636***	***675***	***744***

Note: total road noise cost values presented in *Table 4.3.10*, are based on the sum of the noise costs from highways, regional and city roads.

Road noise damage costs are estimated to be in the region of Euro billion 0.9 in 1995, Euro billion 1.1 in 2010, Euro 1.3 billion in 2020 and Euro billion 1.5 in 2030. Overall road noise accounts for roughly 75% of all transport noise damage in 1995, falling to about 70% from 2010 onwards.

Rail noise damage estimates

Table 4.3.11 presents the total rail noise damage in the Netherlands according to the hedonic property price approach.

Table 4.3.11 Rail noise damage in the Netherlands: Euro million (2000), present value

Noise band: dB(A)	1995	2010	2020	2030
51-55	796.4	1123.0	1141.7	1143.6
56-60	1014.4	1415.9	1533.3	1548.1
61-65	770.7	1025.3	1086.2	1130.7
66-70	456.5	653.2	670.0	659.2
71-75	197.0	315.1	327.9	335.0
76-80	73.8	124.4	122.0	139.8
>80	10.7	24.1	38.6	40.6
Present value	*3320*	*4681*	*4920*	*4997*
Total annual cost	**229**	**323**	**339**	**345**

Rail noise damage costs are estimated to be in the region of Euro million 200 - 300 per annum. Rail noise damage represents roughly 20% of all transport noise in the Netherlands between 1995 and 2020 and this falls to 17% by 2030.

Aircraft noise damage estimates

Table 4.3.12 presents the total aircraft noise damage in the Netherlands according to the hedonic property price approach.

Table 4.3.12 Aircraft noise damage in the Netherlands: Euro million (2000), present value

Noise band	1995	2010	2020	2030
51-55	466.4	966.4	1097.7	1216.0
56-60	363.6	944.9	1207.0	1385.0
61-65	72.8	173.3	252.3	324.6
66-70	17.8	33.3	78.4	89.9
71-75	0.0	8.5	9.1	9.6
76-80	0.0	0.0	0.0	0.0
>80	0.0	0.0	0.0	0.0
Present value	*920.6*	*2126.5*	*2644.5*	*3025.1*
Total annual cost	**63**	**147**	**182**	**209**

Aircraft noise damage in the Netherlands is estimated to be Euro million 63 in 1995, Euro million 147 in 2010, Euro 182 in 2020 and Euro 209 in 2030. Aircraft noise damage represents about 5% of all transport noise in 1995, this increases to about 10% in 2010 to 2030.

4.3.6 Uncertainty

The two main areas of uncertainty in the damage estimates for both the CVM and hedonic property price are the population exposure data to noise and the assumption regarding the threshold for noise (assumed in this study as 50dB(A)). The direction of bias in the results due to the assumption regarding the noise threshold is uncertain because some studies suggest the threshold should be nearer 40dB(A), whilst others favour 55dB(A).

Results based on the unit damage approach may be an underestimate, this is mainly because of the assumption that all noise types are valued the same despite evidence that suggests aircraft and rail noise may be more 'annoying' than road noise.

The main areas of uncertainty in the results based on the hedonic property price approach are:

- noise sensitivity depreciation index (NSDI);
- average house price;
- WTP per excess decibel per person per annum, and
- assumption that all noise types have same impact and are given same values.

The noise damage results based on the hedonic price approach could be an underestimate this is because such an approach values noise only in the context of the home.

The influence of the different variables on the final result is demonstrated in the sensitivity analysis below.

Sensitivity analysis
We conduct a sensitivity analysis of the main areas of uncertainty, identified above. This includes:

Unit damage value approach
- WTP per ‘excess’ decibel per person, per annum, based on contingent valuation approach
- WTP = Euro 15 pp pa pdB(A)
- WTP = Euro 20 pp pa pdB(A)
- WTP = Euro 40 pp pa pdB(A)
- Rising WTP with increasing noise band, see *Table 4.3.7*.

Hedonic property price approach
Sensitivities are tested against the results based on NSDI 4%, noise threshold 50dB(A), average house price Euro 124,921 and the population data as given in *Table 4.3.2*.

- lower and upper NSDI values, i.e. 0.2 - 0.7%;
- noise threshold value, i.e. 55dB(A) based on Maddison et al. (1996), and
- 1999 house price data.

Table 4.3.13 presents the baseline results. The ‘sensitivities’ or changes in the assumptions are then listed and presented with their associated quantitative effects.

Table 4.3.13 Sensitivity analysis: Euro million (2000)

	1995	*2010*	*2020*	*2030*
CLS: Annual noise damage Unit damage value approach, WTP = Euro 20 pp/pa/pdB(A)	1602	1844	1930	2038
Sensitivities				
WTP = Euro 15 pp /pa / pdB(A)	1201	1383	1447	1528
WTP = Euro 40 pp / pa /pdB(A)	3203	3687	3860	4076
WTP based on Bertrand's meta analysis (1997) See *table 4.3.7*	7777	9639	10471	11401
Hedonic property price approach NSDI = 0.4%, noise threshold = 50dB(A), average house price Euro 124,921, population exposure data as in table 4.3.2	1180	1526	1709	1896
(i) NSDI = 0.2%-0.7%	590 - 2064	763 - 2671	854 - 2991	948 - 3319
(ii) noise threshold = 55dB(A)	503	689	793	900
(iv) house price (1999) levels, Euro 157,000	1482	1526	1708	1896

From the sensitivity analysis we conclude that the results are very sensitive to the assumption of a constant linear relationship between WTP and noise levels, i.e. compare the CLS annual noise damage results with those derived from WTP based on Bertrand's meta analysis (1997).

We can also conclude that:

- noise damage values are highly sensitive to the choice of NSDI;
- noise damage values are highly sensitive to the choice of the noise threshold, and
- noise damages based on Eurostat (1995) average house price data are assumed to be an underestimate due to the very low house price reported.

4.4 Land contamination

4.4.1 The issue

Land is contaminated when the concentration of at least one chemical is above the intervention value, which is defined as the level of contamination above which the functioning of humans, animals and vegetation are or will be threatened and hence land may not be permitted for some uses.

4.4.2 Source of emissions

The main categories of contaminated sites include present and former industrial sites, landfills, car dumps and gasworks. The general approach to site 'clean-up' is to maintain the 'multifunctionality' of the soil, i.e. maintain its potential use for various purposes, including agriculture and drinking water supply.

This study does not address the issue of contamination of sediments in the beds of Dutch watercourses and lakes, reported to be a major problem by OECD (1995).

4.4.3 Physical measure of impacts

Table 4.4.1 shows the number of contaminated sites in the Netherlands, their distributions amongst the land use types (1996 figures) and the distribution of land uses when all contaminated sites are cleaned. The table also shows the size (ha) of contaminated sites both for 1996 (all contaminated) and the time when all contaminated land will be cleaned. It is assumed that on average each site is 1.8 ha. The influence this assumption has on the final result is tested in the sensitivity analysis below.

Table 4.4.1 Contaminated areas

	Contaminated Land in 1995		Cleaned Land by 2030	
	No. of sites	Land area (ha)	No. of sites	Land area (ha)
Total	351,000	631,800	351,000	631,800
Residential	91,228	164,210	150,000	269,993
Industrial	126,328	227,390	90,768	163,384
Agricultural	41,084	73,953	39,430	70,970
Recreational and nature	23,292	41,926	26,676	48,017
Derelict	29,600	53,280	4,696	8,453
Dumping	7,925	14,267	2,179	3,924
Other	31,540	56,774	37,248	67,046

Source: RIVM, 1997 Achtergronden bij: Milieubalans 1997, Samson HD, Tjeenk Willink Alhen aan de Rijn.

Not all the sites reported in *Table 4.4.1* are equally contaminated. In fact, the Dutch government already has a classification scheme as follows:

60,000 sites (17% of the total contaminated sites) are 'urgent';
116,000 sites (33% of the total contaminated sites) are 'serious';
175,000 sites (50% of the total contaminated sites) are 'non-serious',
Total sites = 351,000.

However, this distinction is not included in this analysis.

4.4.4 Aggregate monetary damage estimate

Framework for analysis

Baseline	current situation of contaminated land,
Full compliance	all currently contaminated land areas in the Netherlands meet the national standard for clean land.

By comparing the economic value of land in the Baseline scenario to the economic value of land in the Full compliance (FC) scenario, we get an estimate of the benefits of policy measures that ensure full compliance with national standards for clean land.

Changes in the economic value of the land use, benefit of FC scenario, arise for two main reasons, i) changes in the contamination damage within any given land use; and ii) changes in the land use category.

We assume that the use and value of land that currently meets the national standard for clean land is unchanged between the two scenarios and is therefore not included in the analysis.

In the analysis we consider only contaminated land. The surrounding area may also benefit from clean up of contaminated land, this impact is difficult to measure and therefore omitted from the analysis.

Benefits of clean up
Benefits of full compliance (or clean-up) can be defined as the avoided damage (or cost) of contamination. Benefit categories (human health, environmental quality etc) are dependent on the type and level of contamination and the type and amount of environmental (or economic) assets within the contaminated area.

Box 4.4.1 presents the categories of benefits that are relevant to polluters (or private benefits) only as well as benefits that are relevant to the society as a whole.

Box 4.4.1 Suggested benefit categories

Social benefits
- Increased property values which measure the benefit to local people and which may include the health benefits to local people,
- Health benefits to visitors to the area,
- Recreational benefits to local residents in so far as these are not captured in the change in property values,
- Recreational benefits to visitors to the area,
- Reduced ecosystem damage not otherwise captured in recreational or property value increase,
- Gains in 'non-use' value, and
- Gains in 'option' value.

Owner benefits
- Increased property value,
- Elimination of corporate financial environmental liability,
- Elimination of potential for litigation / prosecution,
- Positive public relations value and avoidance of negative public relations, and
- Protection of a resource used as a key input to the problem holder's production process.

Measurement of benefits

Owner benefits are reflected in the land value. It is not possible to measure the social benefits, due to lack of full scientific understanding of the effects of contamination of soils and the site-specific nature of the effects that are understood. Despite this shortfall, land values may include some social benefits, unfortunately it is not possible to determine which social benefits, nor their size. This suggests that change in land value is a conservative estimate of the benefits of clean up.

The present value of benefit of clean up is equivalent to the sum of all land value changes due to clean up. This value will be different for agricultural, residential and industrial / commercial land use types and can be measured by the sale value of land based on the currently available records and views of estate agents, e.g. average price per ha of agricultural land, average value of private housing per hectare.

Application of methodology

Stage 1: estimate economic value of land in the Baseline:	
1	Identify total land area currently defined 'contaminated' **(LB)** *Table 4.4.1*
2	Define current land use (**i**) for land areas defined 'contaminated' i.e. agriculture, housing, industry / commercial, wildlife, derelict. (**LB = Σ Li)** *Table 4.4.1*
3	Identify average land use value for contaminated land, i.e. value for agriculture, housing, industry etc**,** (**Euro per hectare**) Note that property values must also be Euro / ha, (**Pi**) We assume the value for derelict and dumping land is zero. For the baseline calculations it is assumed that the price of contaminated land is 10%less than the price of clean land with all other characteristics remaining the same. For sensitivity analysis, it is assumed that the price reduction due to contamination is 5%.
4	Calculate baseline total value of contaminated land (**B_LD = Σ Pi . Li**) *Table 4.4.2*
Stage 2: estimate economic value of land in the FC	
5	Assume contaminated land area is now cleaned to a 'clean' state, **LS** *Table 4.4.1*
6	Determine possible new land uses (**j**) for land areas cleaned to a 'clean' state. Where (**LS = ΣLj**). *Table 4.4.1*
7	Identify average land use value for 'clean' land, i.e. value for agriculture, housing, industry etc, (**Euro per hectare**) (we assume value of 'clean' land is the same as value of land that has never been contaminated, i.e. full price) Property values must also be Euro / ha (**Pj**) Residential (average of raw lot and ready for building) = Euro607,620/ha Industrial (firms) = Euro586,300/ha Agricultural = Euro21,320/ha Recreational and nature = Euro5,330/ha Derelict and dumping assumed to be zero 'Other' is assumed to equal industrial = Euro586,300/ha
8	Calculate total value of previously contaminated land now cleaned, i.e. (**FCD = Σ Pj x Lj**) *Table 4.4.2*
Stage 3: estimate benefit of FC	
9	Compare land use value in baseline to land use value in FC, i.e. **benefit of FC = FCD - B_LD.** Estimates are present values. *Table 4.4.2*

Table 4.4.2 presents the results of *Stage 2* above, where it is assumed that price of contaminated land is 10% less than the price of equivalent clean land[17]. It is also assumed that all contaminated land is cleaned.

Table 4.4.2 *Benefits of full clean-up PV: Euro million (2000 prices)*

Residential	74254
Industrial	-24195
Agricultural	94
Recreational and nature	55
Derelict	0
Dumping	0
Other	9351
TOTAL	**59559**
Annual value over infinity	3371

The benefits are determined by two main factors:

- change in the value of land, i.e. 10% increase, and
- change in the distribution of the land uses within the contaminated area. The shift is towards residential from other uses. Given that residential area has the highest value, the large magnitude for the benefits of clean-up is reached.

4.4.5 Uncertainty

The main areas of uncertainty in estimating the benefits of cleaning contaminated land are as follows:

- number of contaminated sites: the estimate given in *Table 4.4.1* is probably an underestimate of the current situation. When all potentially contaminated sites are examined (in 2005), it is likely that the total number of sites will increase 1.5 times.

- conversion of 'number of contaminated sites' to size of contaminated land: the average area of a contaminated site (1.8ha) is probably an overestimate. This figures could be closer to 0.94 ha for 'urgent' locations and 0.22 ha for serious locations.

- value of clean land is clearly very much dependent on the location. However, such differences cannot be well represented by the average values used;

- value of contaminated land: According to guidelines for real-estate agencies prices are influenced by contamination in five different ways. These are: cost of clean up (and who has to pay for the clean up), limitations of the use possibilities, inconvenience, negative image of the surroundings and uncertainty. The cost of clean up are subtracted from the price as long as the owner of the land has to pay for the clean up. This means that when the government pays for the clean up operation there are no costs subtracted. When the costs of clean up are for the owner it is important to realise that the value of the land can not be negative. So when the costs of clean up are higher than the value of the land, the value will be zero. For the limitations of the use value there is a subtraction of 5% of the value of the property for houses with a garden. This is because there are limitations in using the garden. There are no subtractions when the land is used industrial. Inconvenience occurs during the clean up operations and therefore 10% of the value is subtracted for residential and industrial use. This subtraction can even be 20% if the inconvenience is extreme. It is important to realise that the percentages are percentages of the

[17] The value of contaminated land at 10% less than uncontaminated land supported by the on going work by EFTEC and others for the Canadian Council of Ministers to the Environment (CCME) and the UK Environment Agency.

total estate value, thus not only the land value but also the buildings on the contaminated land. The negative image of the surrounding has to be concluded from market development because this is different for each location. This is also the case for uncertainty. In other words for uncertainty and negative image no general estimations can be done. In general it can be said that the value of contaminated land highly depends on the location and that it is hard to make general estimates. Because in the past the government did the clean up, there were no transactions of contaminated land. Another problem is that the Dutch housing and industrial offices market is very tight, so it is hard to draw conclusions from the market situation. The only real information that is available is the 5% and 10% subtraction of the property value, as mentioned above.

Overall, the uncertainty of the damage estimates for land contamination in the Netherlands is great. The presence of large uncertainty makes it all the more essential that benefits analysis of land contamination is conducted in future, it serves to both expand knowledge in the area and reduce uncertainty and act as a signal to the Netherlands to collect more relevant data.

Sensitivity analysis

1. The value of contaminated land is 5% less than the value of clean land (baseline assumption is 10% difference);

2. The average area of a contaminated site is 0.58 ha (average of 0.22 and 0.94 ha as mentioned above) (baseline assumption is 1.8 ha per site);

3. Average area of a contaminated site is 0.58ha and the value of contaminated land is 5% less than the value of clean land.

The results of the above sensitivity analyses are presented in *Table 4.4.3.*

Table 4.4.3 Sensitivity analysis: PV: Euro million (2000 prices), annuity at 6%

	1	*2*	*3*
Residential	69265	23927	22320
Industrial	-30861*	-7796*	-9944*
Agricultural	15	30	5
Recreational and nature	44	18	14
Derelict	0	0	0
Dumping	0	0	0
Other	7687	3014	2477
TOTAL	46150	19193	14872
Annual value over infinity	2612	1086	842

* net cost of clean up.

Sensitivity analysis above confirms the importance of the magnitude of the value and size of land. It also shows that both in baseline and sensitivity analysis, clean-up of industrial land (given the assumptions explained above) results in a net cost. It is mainly because due to the high value and area of residential land that clean-up generates net benefits on the whole.

Given the degree of uncertainty associated with the assumptions made in the above analysis, we consider the results for this problem are less certain than the results for the other environmental issues. The assumptions reflect the site-specific nature of the problem rather than an inherent uncertainty in the methodology. In other words, if the same exercise were done for a given site (rather than the whole country), the results would be significantly more robust.

4.5 Particulate matter

4.5.1 The issue

Suspended particulate matter (PM) is made up of a variety of materials and discrete objects and may be liquid or solid, organic or inorganic. PM makes up most of the visible and obvious form of air pollution and is a contributor to summer and winter smogs characteristic of urban areas. Pollutant particles vary in size, from 0.001 μm to 10 μm; small solid particles include carbon black, silver iodide, combustion nuclei and sea salt nuclei. Large particles include cement dust, wind blown soil dust, foundry dust and pulverized coal. Liquid particulate matter or mist, includes raindrops, fog and sulphuric acid mist. Particles of organic origin include viruses, bacteria and spores. The most immediate effect of particles is loss of visibility but particulate matter also influences weather and air pollution phenomena by providing active surface on which heterogeneous atmospheric reactions can occur.

Air pollution can affect human health by damaging the respiratory tract directly or by entering the blood or lymph systems. Soluble particulate matter can also be transported to organs some distance from the lungs. A strong correlation has been found between increases in the daily mortality rate and acute episodes of air pollution. Damage may also occur to buildings, historic monuments and vegetation near or within cities.

4.5.2 Source of emissions

Particles are commonly suspended in the air near the sources of pollution, such as the urban atmosphere, industrial plants, highways, and power plants. The major sources of particulate matter in non-industrial urban areas, are from coal and wood heating and road transport burning diesel fuel. In industrial cities, the main source of particulate matter is power generation.

4.5.3 Physical measure of impacts

Population at risk

The main environmental impacts of particulate matter are the ill-health effects, such as premature mortality and morbidity.

Current legislation scenario: emissions and impacts

The data for impacts associated with exposure to PM_{10} are drawn from the NEO5 (draft report), using the medium scenario (EC). The emissions data for PM_{10} are presented in *Table 4.5.1*, whilst *Table 4.5.2* gives the premature mortality and morbidity cases per annum. The dose / exposure response functions used to establish the human health effects are given in full in the *Appendix* to this chapter.

Table 4.5.1 PM_{10} emissions: 1000 tonnes

	1990	*2010*	*2020*	*2030*
Consumers	6.8	6.5	6.6	6.7
Energy sector	0.6	0.4	0.3	0.3
Industry	14.3	7.9	8.2	8.4
Other	1.3	0.7	0.7	0.2
Refineries	4.8	2.0	1.8	1.8
Transport	18.6	9.9	9.8	11.0
TOTAAL	46.4	27.5	27.4	28.4

Source: NEO5 (draft report).

It is important to note that not all particulate matter is of ‘anthropogenic’ origin. Some arises as a natural background level (e.g. dust) and is not generally subject to policy measures. This study considers the impact of particulate matter of 'anthropogenic' origin only.

Table 4.5.2 Mortality and morbidity effects due to PM_{10}

Mortality (number of people)				
	1995	*2010*	*2020*	*2030*
Total	931	992	1172	1529
Aged under 65	135	147	152	142
Aged 65+	796	845	1020	1387
Morbidity (number of unscheduled hospital admissions)				
Total	2304	2315	2866	3500
COPD	958	953	1161	1401
Asthma	133	122	129	136
Cardiovascular disease	1214	1240	1576	1963

Source: NEO5 (draft report).

Air pollution-related mortality affects largely the elderly, in this study we see that 88% of premature deaths are in the over 65 group. Note that total mortality in *Table 4.5.2* is restricted to mortality caused by cardiovascular disease, COPD and pneumonia[18]. Also note that the morbidity data are restricted to unscheduled hospital admissions for three impacts: COPD, asthma and cardiovascular disease. Exposure to PM_{10} causes other morbidity impacts, which do not result in hospitalisation but which nevertheless should be valued for a complete picture of morbidity effects resulting from PM_{10}. Data are unavailable for the incidence of other morbidity effects, however the sensitivity analysis in the final section explores the effect of different possible assumptions with regard to morbidity.

Particulate matter also has effects on visibility, this is not included in the benefits assessment due to the lack of valuation work in this area.

4.5.4 Monetary measure of impact

Damage estimates due to particulate matter are measured in terms of number of premature deaths and unscheduled ill-health incidences. We can interpret the damage estimates as a measure of the potential benefits that could be secured due to the introduction of measures that control particulate matter, where benefits are understood to be the avoided deaths and ill-health incidences.

Human health

Valuation of morbidity impacts to human health are undertaken using the sum of three values: (i) willingness to pay to avoid each type of episode of ill-health, (ii) productivity loss to employers, and (iii) health care costs of treatment for emergency room visits and hospital admissions, where these are relevant. These values are specific to the Netherlands, and are taken from CSERGE et al., 1999.

Premature mortality impacts are valued using a value of statistical life (VOSL) of Euro million 3.47 in 2000 prices. However, as noted above particulate-related premature mortality affects largely the elderly (over 85% of premature deaths are in the over 65 group). There is some evidence that values of risk aversion are lower for this age group at around 70% of the prevailing risk values (see Pearce 1998). This reduces the VOSL to Euro million 2.4 (2000 prices). The unit values used to value all health impacts are presented in *Table 4.5.3*.

[18] In the Fifth National Environmental Outlook total mortality is defined as all deaths excluding accidents etc.

Table 4.5.3 Unit values of episodes of ill-health and premature mortality (Euro 2000)

Epidemiological end point	Cost
Morbidity effects	
Symptom day	61
Minor restricted activity day	42
Work loss day	117
Respiratory bed day	299
Emergency room visit	559
Hospital admission	1,115
Mortality	
Under 65 years	3,470,000
65 years and over	2,400,000

Applying these values to the morbidity and premature mortality data presented in *Table 4.5.2* gives the damage estimates summarised in *Table 4.5.4* below.

Table 4.5.4 Health damages caused by exposure to PM_{10}: Euro million (2000)

Mortality				
	1995	**2010**	**2020**	**2030**
All ages	2379	2537	2974	3821
Under 65	*468*	*509*	*527*	*492*
65+	*1911*	*2028*	*2447*	*3329*
Morbidity				
Hospital admissions	3	3	3	4
Total	**2382**	**2540**	**2977**	**3825**

Note that these estimates exclude certain effects due to lack of data, either scientific data or willingness to pay data. Damages to visibility and materials are excluded, as are all morbidity impacts which do not result in hospital admissions. For this reason, these figures should be interpreted as an underestimate of the true damage caused by PM_{10}.

4.5.5 Uncertainty

The main areas of uncertainty in the damage estimates due to PM_{10} are:

- relationships between exposure to PM_{10} and premature mortality and ill-health incidences;
- assumption that all fractions of PM_{10} are equally aggressive to human health;
- treatment of premature mortality valuation
- morbidity valuation estimates, and
- omission of impacts to visibility.

Due to the omission of some significant impacts, such as the ill-health / premature mortality incidences due to exposure to $PM_{2.5}$, ill health incidences that do not result in hospital admissions and the impacts to visibility, the damage estimates are considered to be biased downwards.

Sensitivity analysis

We provide a sensitivity analysis of the main areas of uncertainty, identified above. This includes:

- lower and upper bound estimates, based on the 95% confidence interval, for ill-health and premature mortality incidences;

- different values for VOSL;
  - no age adjustment for the premature mortality of people over 65 years;
  - recent CSERGE et al (1999) VOSL with age adjustment, and

- recent CSERGE et al (1999) VOSL without age adjustment.

- different morbidity valuations, based on ExternE values.

Table 4.5.5 presents the adopted assumptions and the baseline results derived. The 'sensitivities' or changes in the assumptions are then listed and presented with their associated quantitative effects.

Table 4.5.5 Sensitivity analysis: Euro million (2000)

Adopted assumption: *premature mortality* Premature mortality (mean estimate) valued with VOSL: Euro million 3.47 for people under 65 years and Euro million 2.4 for those over 65 years.								
	1995		***2010***		***2020***		***2030***	
Current legislation scenario estimate *mean*	2382		2540		2977		3825	
Sensitivities								
• no age adjustment	3231		3440		4065		5304	
• CSERGE VOSL age adjusted	3415		3679		4315		5546	
• CSERGE VOSL no age adjustment	4637		4937		5834		7612	
	lower	upper	lower	upper	lower	upper	lower	upper
Baseline estimate (lower, upper)	79	6770	81	7147	93	8389	125	10523
Sensitivities								
• no age adjustment	102	9065	111	9564	127	11336	175	14470
• CSERGE VOSL age adjusted	107	9315	117	10360	134	12160	181	15270
• CSERGE VOSL no age adjustment	146	13009	159	13726	183	16269	251	20760
Adopted assumption: *morbidity* Morbidity valued based on CSERGE et al (1999)								
Baseline estimate	4		4		5		6	
Sensitivity								
• ExternE values	32		32		39		47	
	lower	upper	lower	upper	lower	upper	lower	upper
Baseline estimate (lower, upper)	1.6	6.9	1.6	6.9	2.0	8.1	2.5	9.7
Sensitivity								
• ExternE values	12.7	54.8	12.8	64.9	16.0	64.9	19.7	77.3

We conclude from the sensitivity analysis the following:

- health damage estimates are highly sensitive to the statistical relationships between exposure to PM_{10} and ill-health and premature mortality incidences;

- health damage estimates due to PM_{10} are sensitive to the manner in which premature mortality is valued and to a lesser extent the approach to morbidity valuation.

Appendix Dose / exposure response functions used

Health Effects

Dose / exposure response functions used in the modelling exercise

The dose-response functions given here are specific to the Netherlands, they are higher than those dose-response functions typically used elsewhere.

Table 4.5.6 Dose-response functions used in this modelling exercise

Mortality: relative risk Relative risk associated with a 80 μg/m³ change in the 24h average PM10 concentration (and 95% confidence interval)				
Age	All causes	Cardiovascular	COPD	Pneumonia
<45	0.927 (0.844 - 1.018)	0.906 (0.728 - 1.128)	1.153 (0.587 - 2.268)	1.427 (0.806 - 2.525)
45-64	1.008 (0.964 - 1.053)	1.023 (0.945 - 1.106)	1.139 (0.841 - 1.541)	1.712 (1.042 - 2.815)
65-74	1.017 (0.979 - 1.056)	1.002 (0.945 - 1.062)	1.166 (0.991 - 1.372)	1.240 (0.879 - 1.748)
75+	1.030 (1.006 - 1.055)	1.016 (0.981 - 1.052)	1.066 (0.965 - 1.178)	1.123 (1.011 - 1.247)
all ages	1.021 (1.002 - 1.040)	1.012 (0.984 - 1.041)	1.099 (1.015 - 1.191)	1.148 (1.040 - 1.268)
Morbidity Emergency hospital admissions RR for a 80 μg/m³ change in the 24h average PM10 concentration (and 95% confidence interval)				
	All respiratory diseases	Chronic obstructive pulmonary disease	Asthma	Cardiovascular Diseases
<15	0.939 (0.896 - 0.984)	0.922 (0.799 - 1.064)	0.895 (0.794 - 1.010)	-
15-64	1.063 (1.015 - 1.113)	1.120 (1.029 - 1.219)	1.140 (1.000 - 1.299)	1.001 (0.970 - 1.034)
65+	1.078 (1.035 - 1.121)	1.160 (1.096 - 1.227)	1.125 (0.897 - 1.409)	1.038 (1.012 - 1.064)
all ages	1.027 (0.998 - 1.056)	1.111 (1.061 - 1.164)	0.995 (0.914 - 1.083)	1.022 (1.001 - 1.043)

Source: daily mortalitiy (acute) from Hoek et al. tables 16a-d, 16i en 19a-d. Morbidity dose response functions are from Vonk and Schouten tables 20a-d.

For comparison, the dose-response functions used in ExternE (1997) are given in *Table 4.5.7*. These have not been used in the sensitivity analysis due to the complexity of the modelling exercise. However, they are provided here for transparency and comparison.

Table 4.5.7 Dose-response functions used elsewhere in the literature

Health impact	*Coefficient*	*Source*
Acute mortality	0.04%	Spix and Wichmann, 1996
RHAs (all population)	2.07×10^{-6}	Ponce de Leon et al., 1996
Cerebrovascular hospital admission	5.04×10^{-6}	
RADs (all population)	0.025	Ostro and Rothschild, 1989

Source: ExternE National Implementation the Netherlands, IVM (1997).

4.6 Eutrophication

4.6.1 The issue

The process of eutrophication is caused by nutrient imbalances, which disturb the natural biochemical balance of ecosystems. In particular eutrophication restricts the intentional uses of water-bodies over large areas. The nutrients of particular concern are nitrogen (from ammonia and nitrate) and phosphorous. The enrichment of natural waters by nutrients, primarily nitrogen in marine waters, but also phosphorous in low salinity waters, has been associated with increased primary productivity and nuisance algal growth in the coastal zones and semi-enclosed areas of seas. The consequences of this can be increased frequency of algal blooms (sometimes toxic), increased water turbidity, slime production, oxygen depletion in deep waters and mass fish and benthic fauna kills. The issue of contaminated groundwater and its health aspects to humans is also a cause of wide concern.

4.6.2 Source of emissions

Domestic sources dominate nitrogen emissions, accounting for approximately 63% of total emissions. This is followed by agriculture (11%) then industrial sources and atmospheric depositions in roughly equal proportions (10% each). The major sources of phosphorous emissions are domestic (e.g. washing powders) and industrial sources, each accounting for approximately 45% of emissions.

4.6.3 Physical measure of impacts

Current legislation scenario: emissions

The data for nitrogen and phosphorous emissions are drawn from the NEO5 (draft report), using the medium scenario ('EC'). These are reported in *Table 4.6.1*. *Table 4.6.1* also gives the sectoral breakdown of emissions for each pollutant.

It is important to note that these emissions relate only to emissions from sources within the Netherlands. The Netherlands is also likely to be affected by emissions from other European countries, for example emissions into surface waters which flow into the Netherlands. Unfortunately, however, data on emissions from other European countries which affect the Netherlands are not available.

The receiving environment is obviously of relevance for the valuation of these emissions. Unfortunately, at this point the existing data are not available at a very disaggregated level. However, RIVM have indicated that approximately 50% of nitrogen is emitted to 'regional waters', defined as ditches, small streams, small canals, with the remainder emitted to 'large waters', i.e. rivers, large canals, lakes etc. For phosphorous, approximately 30% to 40% of emissions are to regional waters.

Table 4.6.1 Nitrogen and phosphorus emissions: current legislation scenario

N (thousand tonnes)				
	1996	*2010*	*2020*	*2030*
Total emissions	56	33	33	33
Domestic effluent	35.06	21.84	22.81	23.46
Agriculture	6.27	0.92	0	0
Industry	5.70	3.80	3.80	3.80
Atmospheric deposition	5.62	4.21	4.01	3.88
Sewage	1.20	0.98	0.93	0.88
Overflow	1.09	0.97	0.88	0.87
Households not connected	0.69	0.23	0.23	0.23
P (thousand tonnes)				
Total emissions	7.6	3.5	3.6	3.6
Industry	3.55	0.66	0.66	0.66
Domestic effluent	3.36	2.64	2.73	2.80
Overflow	0.12	0.09	0.11	0.11
Households not connected	0.11	0.07	0.07	0.07
Agriculture	0.44	0.03	0.01	0.00

Source: NEO5 (draft report)

4.6.4 Monetary measure of impacts

There is a lack of reliable dose-response functions describing the relationships between nitrates, phosphorus and eutrophication. Therefore in order to value the impact of eutrophication it is necessary to consider people's willingness to pay to improve water quality in general.

There is a significant literature on valuation of changes to water quality, which is summarised in *Table 4.6.2.*

Table 4.6.2 Nitrogen and phosphorus emissions: current legislation scenario

Study	*Location*	*Valuation basis*	*Valuation (Euro2000)*
River water quality			
Green and Turnstall (1991)	UK	WTP for improvement in water quality to RE3 WTP for improvement in water quality > RE3	0.94/person/visit 1.10/person/visit
Middlesex University (1994)	UK	Non-use value for improvements in water quality • from very poor to moderate • from moderate to good coarse fishery • from good coarse fishery to trout • from trout to salmon fishery	223,900/km/yr 24,100/km/yr 27,600/km/yr 5,500/km/yr
Hanley (1989)	UK	Guarantee of water supplies with nitrate levels not exceeding 50mg/l	26.66/hh/year
Lant and Roberts (1990)	USA	improvements from poor to fair water quality • recreational value • 'intrinsic' value	39pp.pa 49pp.pa

Table 4.6.2 (ctd) Nitrogen and phosphorus emissions: current legislation scenario

Green and Willis (1996)	UK	WTP of anglers for improvements in water quality • new relatively poor coarse fishery • new good coarse fishery • new good trout fishery Non-use value for improvements in quality • from poor to medium • from medium to good	 6.23/angler/visit 9.95/angler/visit 26.13/angler/vst 0.0087/hh/km/yr 0.0033/hh/km/yr
Reservoir			
Pearson (1992)	UK	Maintain water quality at a standard high enough to support boating and recreational activities	30.20/hh/year
Coastal waters			
Machado et al. (1998)	Portugal	Mean WTP for improvements to 'acceptable' levels of water quality	10.83/person/vis it
Georgiou et al. (1998)	UK	Mean WTP to ensure that EC bathing water standards are met • Great Yarmouth • Lowestoft	 18.60/hh/yr 20.60/hh/yr
Feenberg and Mills (1980)	USA – Boston beaches	WTP for a 10% beach water quality improvement (indicators: oil, total bacteria, colour)	3.50pp.pa (beach users)
Bockstael et al. (1987)	USA – Boston beaches	▪ WTP of users for a 10% beach water quality improvement (indicators: oil, FC, COD) ▪ WTP of users for a 30% beach water quality improvement (indicators: oil, turbidity, FC, COD)	▪ 11.46 pp.pa ▪ 0.57 pp/visit ▪ 36 pp.pa ▪ 1.50 pp/visit
Bockstael et al. (1989)	USA – Chesapeake Bay	WTP for an improvement from 'unacceptable' to 'acceptable' beach water quality	147 pp.pa (beach users)
McConnell and Ducci (1989)	Barbados beaches	WTP for sewerage reduction	10 – 165 pp.pa
McConnell and Ducci (1989)	Uruguay beaches	WTP for sewerage reduction to improve water quality to swimming levels	13 pp.pa
Choe et al. (1996)	Philippine beach	WTP for sewerage reduction to improve water quality to swimming levels	11 – 22 pp.pa
Georgiou et al. (1996)	UK beaches	Improvement of water quality to meet EC standards	18 – 69 pp.pa

One major research project has conducted extensive valuation studies of eutrophication in the Baltic Sea (Turner et al., 1995, 1997). The study involved several contingent valuation and travel cost studies in the context of an assumed 50% nutrient load reduction programme which can legitimately be regarded as a 'maximum feasible reduction' scenario since such a programme has substantial costs of nearly Euro billion 4 per annum. The results are set out in *Table 4.6.3*. The Polish CVM studies involved a beach survey (Zylicz, 1995a) and a household survey, with, in each case, respondents being asked their WTP for clean-up programme (Zylicz, 1995b). The first two surveys give very close results, but the surprising feature is the sheer size of the WTP figures. Taking the Swedish results as being typical of the west European economies and the Polish results as being typical of the EITs, Turner et al. (1997) estimate Baltic basin wide benefits from a clean-up programme to be some SEK69 billion per year, or some Euro billion 8.0 per annum compared to costs of Euro billion 3.6 pa. If non-respondents are treated as having implicit zero WTP for the clean up programme, then benefits reduce to SEK billion 37.9 per annum or about Euro billion 4.4: benefits are still greater than costs though by a fairly narrow margin.

Table 4.6.3 Median Willingness to Pay to Improve the State of the Baltic Sea (2000 Euro)

Study	Country	WTP	% Income
Zylicz et al., 1995a Beach survey, CVM	Poland	62 – 116 pp.pa	3-5%
Zylicz et al., 1995b household survey, CVM	Poland	28 – 69 pp.pa	1.3-3.3%
Mail survey, CVM	Poland	99 – 200 pp.pa	4.6-9.5%
Sandstrøm, 1995, TCM beach use	Sweden	30 per trip	
Søderqvist, 1995, CVM	Sweden	315 pp.pa	

Total N and P loads to the 'Baltic proper' (the geographical zone used in the study) in 1993 were 860,000 tN and 33,000 tP per year (Turner et al., 1995). Based on the total benefit of 50% reduction in this load, *Table 4.6.4* presents the benefits per unit of N and P.

Table 4.6.4 Benefits of nitrogen reduction in the Baltic Sea (Euro/tonne)

Pollutant	Euro/tonne
N	11,030
P	11,120

Source: Authors.

The difficulty in transferring the values produced in any of the above studies to value eutrophication of inland waterways in the Netherlands is threefold, these are:

First, the valuation studies in existence tend to estimate WTP for a given improvement in water quality. It is difficult to estimate the relationship between a change in emissions and a change in water quality since this is highly dependent on the receiving environment. While it might be possible to estimate the relationship on a site-specific basis, it is evidently a much more complex task at a national level.

Second, willingness to pay for water quality improvements will depend on both the existing water quality, and the proposed improvements. There is no satisfactory data available concerning the existing water quality of inland and coastal waters in the Netherlands at the sort of level of detail required for this analysis. Site-specific characteristics of the water body will also be relevant, for example the extent to which it is used for recreational and commercial purposes, whether it is urban or rural, its 'uniqueness' in terms of biological or other functions. It is possible to adjust for these factors on a local scale, when considering changes to a single water resource. However, it becomes much more difficult when considering the national water resources as a whole.

Finally, all of the valuation studies to date have examined WTP for a change to a given water resource. It is generally accepted that WTP for improvements is dependent on the existence of other similar resources ('substitute resources') nearby. Therefore, even if data were available on existing water quality, possible improvements linked to emissions reductions, and site-specific characteristics for all water bodies in the Netherlands, it is not clear that aggregation of WTP for improvements over all water bodies would be valid.

Another possible approach is to use the cost of clean-up for these pollutants as a measure of damages i.e. the cost of controlling discharges is used as a proxy for the damage done by the discharges. The analytical foundations of this approach are far less robust, since it effectively means that control costs equal damage costs, implying that the ratio of benefits to costs of control are always unity. In addition, clean-up cost will vary depending on the technology used. These results should therefore be viewed as supplementary to the WTP results. If the pollutants are actually removed in the treatment process, then cost of clean-up is a true estimate of the damage caused. However, if they are not removed, then willingness to pay estimates are the correct measures of damage.

For the purposes of estimating clean-up costs, it is assumed that the treatment is chosen so that the treated sewage meets the relevant standards across Europe, which are assumed to be based on correct assumptions regarding the assimilative capacity of the receiving environment. A paper by Odegaard (1996) gives the efficiency of each sewage treatment in terms of expected removals of specific nutrients (including total P-phosphate, total N) and the costs of these treatments. The ‘best’ method is ‘advanced tertiary’, using chemical/biological means which remove 95% BOD, 90% total P and 70% total N at a cost of 3.11 NKr/m^3 (0.34 Euro/m^3). More usefully, the cost is given as a cost/efficiency factor, at a cost per kilogram removed. The results are given in *Table 4.6.5* below.

Table 4.6.5 Cost-efficiency factors for chemical and biological pollutants: post-DN, pre-precip

Pollutant	% removed	NKr/tonne	Euro/tonne removed
Total N	70	112,000	15,000
Total P	90	562,000	74,000

For the purposes of the damage assessment, both the cost of clean-up values and the WTP data are used, in order to give a range of possible values for this environmental problem. It should be noted that these values are only indicative of the likely magnitude of the problem, for all of the reasons discussed above.

4.6.5 Aggregate monetary damage estimate

Combining the emissions data of *Section 4.6.3* with the valuation estimates in *Section 4.6.4*, the damage estimates given in *Table 4.6.6* below are obtained.

Table 4.6.6 Damages due to nitrogen and phosphorus emissions (million Euro)

	1996	*2010*	*2020*	*2030*
N	618 - 840	364 - 495	364 - 495	364 - 495
P	85 - 562	39 - 259	40 - 266	40 - 266
Total	702 - 1,402	403 - 754	404 - 761	404 - 761

Note: lower values assume WTP, upper values assume ‘costs of clean up’.

4.6.6 Uncertainty

The main sources of uncertainty are due to the scientific data with respect to the water quality in the Netherlands and the lack of evidence of a WTP for a reduction of eutrophication impacts for inland waters in the Netherlands.

Due to the lack of information regarding the issue of eutrophication we are unable to conduct a meaningful sensitivity analysis.

In order to assess eutrophication in the Netherlands in the future it could require an original contingent valuation survey asking respondents for their WTP for an increase in their local water quality as well as a maximum WTP to increase water quality for all waters in the Netherlands. The results of such an analysis could then be aggregated across the Netherlands in order to establish the overall damage due to eutrophication.

4.7 Low level ozone

4.7.1 The Issue

Low level ozone is caused by emissions of nitrogen oxides and volatile organic compounds reacting in sunlight. Generally low level ozone concentrations are higher during the day and in the summer. However, the issue of estimating the effects of low level ozone is further complicated because it forms over time and may be worse in rural areas downwind of significant sources of emissions. Low level ozone is considered to cause harm to humans via the respiratory organs and the eyes. Ozone also causes damage to vegetation and crops in and around urban areas.

4.7.2 Source of emissions

Low level ozone is a secondary pollutant caused by the interaction of the precursors, nitrogen oxides, NO_x, and volatile organic compounds VOCs, in the presence of sunlight. SO_x and NH_3 are also implicated. Low level ozone is often called photochemical smog or just smog. Due to the role of sunlight, the concentrations of low level ozone are generally higher during the day and in the summer.

Motor vehicles account for a considerable proportion of the total emissions of nitrogen oxides in Europe, and their contribution is expected to increase following the growth in use of the private car. The main source of VOCs is from the combustion of petrol and diesel in urban areas. Other sources include the burning of coal or wood for heating, solvents and industry.

4.7.3 Physical measure of impacts

Estimating damages from low level ozone is complex because it forms over time and may be worse in rural areas downwind of significant sources of emission. Tropospheric ozone is implicated in the forms of damage summarised in *Table 4.7.1*.

Table 4.7.1 Environmental impacts of low-level ozone

Receptor	*Impact*
Human health	Mortality; Respiratory hospital admissions; Restricted activity days; Symptom days;
Crops	Reduced yield;
Forests	Reduction in tree growth; Non-crop vegetation damage, and
Materials	Damage to some materials (paints, plastics, rubbers, metals).

Rabl and Eyre (1997) suggest that while the effect of ozone to materials is potentially significant, it is likely to be small. Thus, low level ozone damage to materials is excluded. Low level ozone is known to reduce tree growth. However, there is currently very little information about the damage to forests due to the complexity of their growth and management systems. Similarly, while it is likely that ozone causes damage to ecosystems, there are very few studies estimating these effects. Thus ozone damage to forests and biodiversity are also excluded from this study.

Effects on human health and crops are analysed below. Recommended D/ERFs for both human health effects and effects on crops are given in full in the *Appendix* to this chapter.

Current legislation scenario: emissions and impacts

The data for the precursor pollutants are drawn from the NEO5 (draft report), using the medium scenario ('EC'). The emissions data for NO_x are presented in *Table 4.2.2* in *Section 4.2*. Data on VOC emissions are drawn from the same source and are presented in *Table 4.7.2* below. *Table 4.7.2* also gives the sectoral breakdown for VOCs.

Table 4.7.2 VOC emissions: current emissions scenario: 1,000 tonnes

Source	*1995*	*2010*	*2020*	*2030*
Transport	148.3	51.9	48.4	52.4
Industry	80.9	55.4	59.5	63.6
Households	39.6	31.1	33.7	37.7
Services	29.2	18.7	20.8	23.1
Energy	26.2	15.8	13.8	13.8
Construction	21.9	10.5	11.8	12.1
Refineries	11.7	9.0	9.0	9.0
Other	4.0	2.4	2.2	2.1
Total	*361.8*	*194.9*	*199.2*	*213.7*

Source: NEO5 (draft report).

Physical damages should ideally be attributed back to the precursor emissions of NO_x and VOCs. However, because of ozone chemistry, it is very difficult to report average damage estimates per tonne of the precursor pollutants. Actual values will vary considerably from place to place, and will be dependent on factors such as temperature and sunlight. This needs to be taken into account in any analysis relevant to a smaller area.

Given the high importance of the location of emissions in ozone chemistry, it is important that damage estimates specific to the Netherlands are used, as these are likely to differ considerably from the European average. Model simulations have therefore been used to produce estimates of the human health impacts (mortality and morbidity) due to low level ozone. These are presented in *Table 4.7.3.* The drawback of this approach is that it is more difficult to relate damages to the sources of emissions. However, it is likely to generate a more accurate estimate of the magnitude of the problem.

Table 4.7.3 Mortality and morbidity effects due to ozone

Mortality: number of people				
	1995	*2010*	*2020*	*2030*
Total	1,540	2,078	2,500	3,246
Aged under 65	110	147	153	141
Aged 65+	1,448	1,962	2,354	3,226
Morbidity: number of unscheduled hospital admissions				
Total	449	619	749	869
COPD	281	389	472	548
Asthma	0	0	0	0
Cardiovascular d.	168	230	278	320

Source: NEO5 (draft report).

Note that total mortality in *Table 4.7.3* is restricted to mortality caused by cardiovascular disease, COPD and pneumonia[19]. Also note that the morbidity data is restricted to unscheduled hospital admissions for three impacts: COPD, asthma and cardiovascular disease. Exposure to ozone also causes other morbidity impacts, which do not result in hospitalisation but which nevertheless should be valued for a complete picture of morbidity impacts resulting from ozone. Data are unavailable for the incidence of other morbidity effects, however the sensitivity analysis of the final section explores the effect of different possible assumptions about morbidity.

For valuation of effects to crops, the relevant data are ozone concentrations exceeding AOT40. These data are presented in *Table 4.7.4* below. This measure is defined for areas of crops and natural vegetation over the three months of May, June and July, which is the main growing season.

[19] In the Fifth National Environmental Outlook total mortality is defined as all deaths excluding accidents etc.

Table 4.7.4 Ozone concentrations exceeding AOT40: ppm.hour

	1995	*2010*	*2020*	*2030*
AOT40 sum averaged over Dutch agricultural area	7.5 – 9.1	7.5 – 9.1	7.1 – 8.6	7.1 – 8.6

4.7.4 Monetary measure of impacts

Human Health

Valuation of morbidity impacts to human health is undertaken using the sum of three values: (i) willingness to pay to avoid each type of episode of morbidity, (ii) productivity loss to employers, and (iii) health care costs of treatment for emergency room visits and hospital admissions, where these are relevant. These values are specific to the Netherlands, and are taken from CSERGE et al., (1999).

Premature mortality impacts are valued using a VOSL of 3.47 million Euro, adjusted to 2.4 million Euro for people aged over 65. The basis of VOSL value is explained in *Annex A3 on Valuing Risk of Premature Mortality*. The unit values used to value all health impacts are presented in *Table 4.7.5* below.

Table 4.7.5 Total costs of episodes of ill-health: Euro (2000)

Epidemiological End Point	**Cost**
Morbidity effects	
Symptom day	61
Minor restricted activity day	42
Work loss day	117
Respiratory bed day	299
Emergency room visit	559
Hospital admission	1,115
Mortality	
Under 65 years	3,470,000
65 years and over	2,400,000

Applying these values to the morbidity data presented in *Table 4.7.3* gives the damage estimates summarised in *Table 4.7.6* below.

Table 4.7.6 Health damages caused by low-level ozone: million Euro (2000)

	1995	*2010*	*2020*	*2030*
Mortality				
All ages	3,858	5,218	6,182	8,232
under 65	*383*	*509*	*532*	*489*
65+	*3,475*	*4,709*	*5,651*	*7,743*
Morbidity				
Hospital admissions	0.5	0.7	0.8	0.9
Total	3856	5219	6183	8233

Crops

Crop damage due to exposure to ozone levels is a function of ozone dose, crop species, cultivar, biological conditions, climatic conditions, soil conditions, production and other factors. Interaction of these variables makes accurate crop loss assessment especially difficult over large areas. The estimates provided below are a first order approximation of crop damage in the Netherlands due to low level ozone.

Ozone is considered to cause yield reduction for 'ozone sensitive' crops. Jones et al. (1997) list these crops as summarised in *Table 4.7.7*.

Table 4.7.7 Ozone sensitive crops

Slightly Sensitive Crops	Sensitive Crops			Very Sensitive Crops
millet	apples	hops	plums	Tobacco
oats	beans	lemons	potato	
pasture grass	carrots	limes	rape seed	
rice	clover	linseed	sesame seeds	
rye	cucumbers	melons	soybeans	
sorghum	dates	oil seeds	sunflower	
	flax	onions	tangerines	
	grapefruit	oranges	tomato	
	grapes	peaches	watermelons	
	hemp	pears	wheat	

Production levels for the relevant crops in the Netherlands are taken from the FAO Statistical Database online. These are presented in the *Appendix* to this chapter. The reduction in yields caused by exposure to ozone is valued at world prices, also taken from the FAO Statistical Database. These represent the true value of crops and are unaffected by market distortions such as subsidies under the Common Agricultural Policy which tend to affect prices significantly.

To obtain an estimate of total damages to crops from low-level ozone, the ozone concentration data given in *Table 4.7.4* have been used in conjunction with the dose response functions listed in the annex plus the FAO production and price data for the crops above. Total damages to crops are summarised in *Table 4.7.8* below.

Table 4.7.8 Crop losses caused by low-level ozone: million Euro (2000)

	1995	*2010*	*2020*	*2030*
All crops	169 – 205	169 – 205	160 – 194	160 – 194

A recent Dutch study, (Tenneijck et al., 1998) supports the crop damage estimates in *Table 4.7.8.* The Dutch study estimates the benefits to the Netherlands in terms of avoided crop damage due to a 70% reduction of 1995 low level ozone concentrations is about Euro 200 million per annum.

The above analysis of crop damage due to ozone exposure is a first order approximation of the true damage. The analysis assumes constant crop prices, even though with reduced ozone levels production could increase forcing prices down. We also assume that the crop mix through time remains the same despite the decrease in ozone emissions. It is likely, that as ozone levels fall there could be a significant reduction in the production of ozone tolerant crops because of the reduced profitability relative to crops that were more sensitive to ozone.

4.7.5 Aggregate monetary damage estimate

Aggregating the damages to human health (mortality and morbidity) and the crop losses estimated in *Section 4.7.4* above, gives estimates of total damages caused by low-level ozone as summarised in *Table 4.7.9* below.

Table 4.7.9 Total damages caused by low-level ozone: million Euro (2000)

	1995	*2010*	*2020*	*2030*
Human health				
Mortality (all ages)	3,858	5,218	6,182	8,232
Under 65	*383*	*509*	*532*	*489*
65+	*3,475*	*4,709*	*5,651*	*7,743*
Morbidity:				
Hospital admissions	0.5	0.7	0.8	0.9
Crop losses				
All crops	169 – 205	169 – 205	160 – 194	160 – 194
Total				
All costs	**4,028 – 4,064**	**5,388 – 5,424**	**6,343 – 6,377**	**8,393 – 8,427**

Note that these estimates exclude certain effects due to lack of data, either scientific data or willingness to pay data. Damages to materials, forests and ecosystems are all excluded, as are all morbidity impacts which do not result in hospital admissions. For this reason, these figures should be interpreted as an underestimate of the true damage caused by low-level ozone.

Damages due to ozone in the Netherlands are expected to increase over time, from Euro 4.1 billion in 1995 to over Euro 8 billion in 2030. This, however, is due to the changing age structure of the population in the Netherlands, which has a growing proportion of the population in the over 65 age groups. The health impacts of ozone affect the elderly disproportionately, and the health impacts account for by far the largest proportion of total damages. Actual ozone concentrations in the country are expected to *decrease* during the time frame under consideration.

It should be noted that the estimates presented here are restricted to the ozone problem *as perceived by the population of the Netherlands*. In other words, some of the ozone formation in the Netherlands will be due to the emissions of NO_x and VOCs from other countries. Also, the emissions of NO_x and VOCs in the Netherlands will tend to contribute to ozone formation outside the country, this damage is not accounted for in the current estimates.

4.7.6 Uncertainty

The main sources of uncertainty in this analysis are:

- approach to premature mortality;
- omission of impacts to materials, forests, ecosystems, non-crop vegetation and biodiversity. Ozone is known to damage some polymeric materials such as paints, plastics and rubbers as well as having a corrosive effect on metals. However, ExternE (CEC, 1995) suggests that further research is necessary before these impacts can be quantified. Similarly, recent evidence suggests ozone is the primary cause of damage to forests and that it causes damage to ecosystems. These effects have all been omitted from the analysis due to uncertainty in both the scientific and economic valuation estimates. Their exclusion will tend to bias the results downward;
- morbidity effects other than hospital admissions – i.e. emergency room visits, symptom days and 'restricted activity days'. All of these are known to be linked to ozone, however, at the time of writing, data on their incidence in the Netherlands were not available, and
- dose-response functions for crops do not take into account of changing farm practices that reduce the susceptibility of crops to ozone.

Overall, the damage estimates for low level ozone may be biased downwards due to the omission of a number of potentially important impacts.

The sensitivities considered are as follows:

1. *Effect of assuming different values for a VOSL*. The valuation of mortality effects is conducted using a 'value of statistical life'. For more details on this approach see *Annex 3 on valuing the risk of premature mortality*. Mortality incidence in the over 65 age group is valued at 70% of a VOSL, based on the results of Pearce (1997). The sensitivity explores three different assumptions about the value of a VOSL, namely:

- the effect of removing the age adjustment;
- using the latest VOSL estimate by CSERGE et al (1999) with age adjustment, and
- using the CSERGE estimate without age adjustment.

2. *Effects of adopting older unit value estimates for valuation of morbidity effects*: The baseline results are based on the latest unit damage estimates from CSERGE et al. (1999). These are the most up-to-date estimates of WTP to avoid these impacts, and are also specific to the Netherlands. For these reasons, the damage estimates produced are likely to be much more accurate than previous estimates. However, most cost-benefit assessments to date have been based on a different set of unit values, adjusted from American studies. These are summarised in *Annex 4 on Monetary Valuation of Health Effects*. Therefore, for purposes of comparison of our results with previous studies, we explore the effect of using the older set of estimates.

The results of these sensitivities are presented in *Table 4.7.10* below.

Table 4.7.10 Sensitivity analysis: million Euro (2000)

	1995	*2010*	*2020*	*2030*
Mortality costs				
Current legislation scenario estimate	3,858	5,218	6,182	8,232
Sensitivities:				
(i) No age adjustment	5,407	7,317	8,701	11,684
(ii) CSERGE value age adjusted	5,603	7,578	8,980	11,961
(iii) CSERGE no age adjustment	7,760	10,501	12,488	16,769
Morbidity costs				
Hospital admissions - baseline	0.5	0.7	0.8	0.9
Sensitivities:				
(i) ExternE values	4.0	5.5	6.6	7.7

Finally, one more sensitivity is conducted based on the scientific uncertainty surrounding the morbidity and mortality impacts:

Upper and lower bounds due to scientific uncertainty: The 95% confidence intervals for mortality and morbidity incidence resulting from the scientific analysis has been combined with the baseline assumptions, and the sensitivities outlined above, to produce an overall range of estimates. *Table 4.7.11* presents the results.

Table 4.7.11 Sensitivity- mortality and morbidity effects due to ozone (million Euro)

	1995		*2010*		*2020*		*2030*	
Mortality costs								
	lower	*upper*	*Lower*	*upper*	*lower*	*upper*	*lower*	*upper*
Baseline	2,231	6,628	3,030	8,845	3,615	10,499	4,987	13,416
Sensitivities:								
(i) No age adjustment	3,221	9,063	4,375	12,113	5,221	14,477	7,205	18,747
(ii) CSERGE value age adjusted	3,244	9,615	4,405	12,833	5,257	15,238	7,252	19,482
(iii) CSERGE no age adjustment	4,622	13,007	6,278	17,384	7,493	20,777	10,341	26,905
Morbidity costs								
Hospital admissions: baseline	0.0	1.7	0.1	2.2	0.1	2.6	0.1	3.0
Sensitivities:								
(i) ExternE values	0.1	14.3	0.5	18.1	0.6	21.7	0.7	24.9

Appendix Dose / exposure response functions assumed

Health Effects

Dose / exposure response functions used in the modelling exercise

The dose-response functions given here are specific to the Netherlands, and are higher than those dose-response functions typically used elsewhere.

Table 4.7.12 Dose-response functions used in this modelling exercise

Acute mortality: *Relative risk associated with a 150 μg/m3 change in the 8h average Ozone concentration between 12:00-20:00 (and 95% confidence interval)*				
Age	All causes	Cardiovascular	COPD	Pneumonia
<45	0.983 (0916 - 1.054)	1.022 (0.863 - 1.212)	2.137 (1.258 - 3.630)	1.142 (0.682 - 1.911)
45-64	1.040 (1.004 - 1.077)	1.038 (0.977 - 1.103)	1.083 (0.856 - 1.371)	0.677 (0.424 - 1.081)
65-74	1.030 (1.0 - 1.061)	1.005 (0.960 -1.053)	0.989 (0.862 - 1.135)	1.597 (1.164 - 2.190)
75+	1.043 (1.024 - 1.063)	1.073 (1.044 - 1.102)	0.996 (0.919 - 1.079)	1.367 (1.245 - 1.501)
All ages	1.039 (1.024 - 1.054)	1.055 (1.032 - 1.079)	1.006 (0.942 - 1.074)	1.146 (1.062 - 1.236)
Morbidity: emergency hospital admissions Coefficient for a 150 μg/m3 change in the 8h average Ozone concentration between 12:00-20:00 (and 95% confidence interval)				
Age	All respiratory diseases	COPD	Asthma	Cardio vascular diseases
<15	0.954 (0.913 - 0.997)	0.838 (0.731 - 0.960)	0.918 (0.834 - 1.010)	-
15-64	1.023 (0.985 - 1.061)	0.925 9 0.855 - 1.001)	1.096 (0.990 - 1.203)	0.993 (0.970 - 1.018)
65+	1.045 (1.008 - 1.082)	1.115 (1.054 - 1.179)	1.051 (0.897 - 1.231)	1.010 (0.990 - 1.029)
All ages	1.017 (0.992 - 1.042)	1.034 (0.989 - 1.082)	1.029 (0.968 - 1.094)	1.002 (0.986 - 1.018)

Source: daily mortality from Hoek et al. Tables 16a-d, 16i en 19a-d. Relative risk for morbidity from Vonk and Schouten tables 20a-d.

For comparison, the dose-response functions used in ExternE (1997) are given in *Table 4.7.13*. These have not been used in the sensitivity analysis due to the complexity of the modelling exercise. However, they are provided here for transparency and comparison.

Table 4.7.13 Dose-response functions used elsewhere in the literature

Health impact	Coefficient	Source
Acute mortality	0.059%*	Sunyer et al.,1996
RHAs (all population)	7.09×10^{-6}	Ponce de Leon et al., 1996
Symptom days (all population)	0.033	Krupnick et al., 1990
RADs (all population)	9.76×10^{-3}	Ostro and Rothschild, 1989

Source: ExternE National Implementation the Netherlands, IVM (1997)

The exposure response coefficients presented in *Table 4.7.13* are for Western Europe and has units of (cases / year-person-µg/m^3) for morbidity and (% change in annual mortality rate/µg/m^3) for mortality.

Further information on the unit values used for the valuation of morbidity and mortality effects can be found in *Annex 4 on Monetary Valuation of Health Effects* since these effects are common to several pollutants.

Crops

Jones et al (1997) reviews the effects of air pollution on crops and reports the dose response functions reproduced in *Table 4.7.14*.

Table 4.7.14 Physical measure of the effects of ozone on crops

Type of crop	Dose-response function
Tolerant crops	0% loss in yields per ppm hour AOT40
Slightly sensitive crops	1.0% loss in yields per ppm hour AOT40
Sensitive crops	1.75% loss in yields per ppm hour AOT40
Very sensitive crops	3.57% loss in yields per ppm hour AOT40

The problems with these DRFs include:

- the omission of farmer adaptation, and
- part of damage that is accounted for by plant adaptation. Ozone concentrations are highest on hot dry days and there is evidence to suggest that plants protect themselves on such days to conserve moisture. This protection also has the effect of protecting against damage from ozone.

These effects are difficult to quantify. However, both would suggest that the use of the dose-response factors given in *Table 4.7.14* without adjustment to account for these effect would result in an overestimate of the effects of ozone on crops. However, without further information on the likely magnitude of this effect, it is impossible to quantify in the estimates presented here.

In order to estimate the effects on production, and the value of crop losses, production and price data for these crops in the Netherlands was taken from the FAO Statistical Database. These are presented in *Table 4.7.15*.

Table 4.7.15 Crop Production in the Netherlands and Prices (1995)

Type of crop	Production (Mt)	Price (1990 $US/Mt)
Slightly sensitive crops		
Rye	42,500	107
Oats	15,500	109
Sensitive Crops		
Apples	560,000	307
Beans, Dry	5,000	539
Beans, Green	72,800	427
Carrots	368,900	163
Cucumbers and Gherkins	506,500	223
Flax Fibre and Tow	34,000	961
Grapes	300*	340
Linseed	6,000	267
Onions, Dry	438,600	202
Peaches and Nectarines	12*	369
Pears	140,000	335
Plums	6,600	309
Potatoes	7,340,000	110
Rapeseed	5,000	328
Tomatoes	560,700*	192
Wheat	1,167,000	144

* data from 1994 since 1995 data not available,
Source: FAO Statistical Database.

Materials damage
Ozone is known to damage some polymeric materials such as paints, plastics and rubbers as well as having a corrosive effect on metals (Lee et al., 1995). However, ExternE (CEC, 1995) suggests that further research is necessary before these impacts can be quantified. The UN ECE Programme (Kucera, 1994) states that the corrosive effect of ozone on metals is very uncertain, but recent evidence shows that ozone is probably important in accelerating some reactions. Kucera (1994) shows that ozone also acts synergistically with SO_2 especially for zinc. Although zinc is not an important construction material, it is used extensively as a protective coating for steel.

The DRFs in the existing literature are presented in *Table 4.7.16* below. These have not, however, been used in the current analysis due to the uncertainty surrounding these estimates, and the lack of a measure of ‘stock at risk’ for the Netherlands. They are presented here for completeness.

Table 4.7.16 Dose-response functions for the effect of ozone on zinc

Type of zinc	Dose-response function (mass loss after 4 years (g/m^2))
Unsheltered zinc	$14.5 + 0.043 \times TOW \times SO_2 \times O_3 + 0.08 \times H^+$
Sheltered zinc	$5.5 + 0.013 \times TOW \times SO_2 \times O_3$

TOW is the time of wetness and H^+ is a measure of acidity ($meq/m^2/year$).
Source: Kucera (1994).

Forest damage
Ozone is the primary cause of damage to forests. At the UNECE Kuopio workshop, the following two dose-response functions were proposed for ozone effects on forest productivity:

Beech: % productivity change = -0.27x
Norway spruce: % productivity change = -0.18x

Where, x is the ozone expressed as AOT40, that is the ozone concentration accumulated over a threshold of 40ppb in daylight hours over the growing season, expressed in ppm.hours. However, there is a high level of uncertainty associated with there two functions, which increases in the extrapolation from two species to all deciduous and coniferous species, and to the whole of Europe (AEA, 1998a). This is compounded by difficulties in monetary valuation. Due to the high levels of uncertainty with these results, valuation of forest damages is not estimated here.

Ecosystem damage

Similarly, while it is likely that ozone causes damage to ecosystems, there are very few studies estimating these effects. They have therefore been omitted from the current analysis.

4.8 Damage assessment and priority issues

4.8.1 Ranking environmental issues according to damage estimates

Ranking environmental issues is useful in the sense of highlighting priority issues and pointing out any surprises, for example, noise nuisance is a priority issue, whereas acidification is becoming less of an environmental problem than it used to be. Such exercises can be used for awareness raising for decision makers. However, ranking alone does not answer any questions about policy. In order to do so we would need to compare the benefits of environmental control with the costs, the unit damage values used in this benefit assessment study can be re-used if a CBA is conducted in future.

Table 4.8.1 summarises the damage assessment results for the various environmental issues in the Netherlands. They are the mean per annum damage estimates and at this stage they are not discounted (with the exception of the damage estimates for land contamination, which are present values of total damage). All values are in Euro (2000 prices).

Table 4.8.1 Total annual damages: not discounted: Euro million, (2000 prices)

	1995	***2010***	***2020***	***2030***
Climate change	1942	1948	2064	2234
Acidification	2647 - 2975	1459 - 1631	1437 - 1610	1516 - 1700
PM10	2382	2540	2977	3825
Eutrophication	702 - 1402	403 - 754	404 - 761	404 - 761
Low level ozone	4028 - 4064	5388 - 5424	6343 - 6377	8393 - 8427
Noise	1602	1844	1930	2038
Land contamination *Present value*	59559			

Due to the existence of international treaties for the control of climate change and acidification, the climate change estimates are damages to the world due to greenhouse gases from the Netherlands, and for acidification the estimates are damages to the UNECE due to acidifying pollutant emissions from the Netherlands. Note that the land contamination damage estimates are present values.

In order to prioritise the environmental issues in order of damages (or mean potential primary benefits from control), the damage estimates must be made directly comparable. This is complicated by the fact that damage estimates for land contamination are already present values. In order to include the land contamination damage assessment in the comparison with the other environmental issues we convert all damage estimates into present values (discount rate = 6%). *Table 4.8.2* gives the total damage estimates as a present value as well as the corresponding annual damage value (for the relevant formulas refer to *Definitions*). It is now possible to compare and rank the environmental issues in the Netherlands in terms of greatest potential primary benefits from control. *Table 4.8.2* gives the ranking.

Table 4.8.2 Total and annual damage estimates for environmental issues in the Netherlands

	total damage (mid) present value discount rate=6% Euro million (2000)	Annual damage (mid) Euro million (2000)	Ranking	Qualitative assessment
Low level ozone	110034 - 110613	6228 - 6261	1	++
Land contamination	59559	3371	2	--
PM10	54471	3083	3	++
Acidification	37017 - 41569	2095 - 2353	4	++
Climate change	36766	2081	5	++
Noise	31980	1810	6	+
Eutrophication	9835 - 19224	557 - 1088	7	--

Note: the annual damage values assume a linear distribution of benefits between 1995-2030.

Thus we see that the greatest priority, in terms of potential primary benefits from control, is to low level ozone, land contamination and particulate matter, followed by acidification and climate change. Whilst eutrophication and noise are estimated to yield the lowest potential benefits from control. *Table 4.8.3* shows how the rankings change if we consider the upper or lower annual damage estimates. For example, rankings based on upper estimates suggest the greatest priority, in terms of potential primary benefits from control is to, low level ozone, PM_{10}, and climate change, followed by noise nuisance, land contamination, acidification and eutrophication. In other words, low level ozone and eutrophication maintain their original rankings as top and least priority, whilst PM_{10}, climate change and noise nuisance move up in terms of importance and land contamination and acidification fall. Rankings based on the lower damage estimates again suggest that low level ozone and eutrophication are top and low priority respectively, interestingly, PM_{10} and land contamination both fall to become low priority issues, whilst acidification and noise nuisance increase in importance.

Table 4.8.3 Annual damage estimates for environmental issues in the Netherlands: lower / upper estimates. Euro million (2000 prices)

	annual damage:lower Euro million	ranking based on lower estimates	annual damage: upper	Ranking based on upper estimates
Low level ozone	3793	1	10219	1
PM10	101	7	8584	2
Climate change	624	5	4647	3
Noise nuisance	1357	3	3620	4
Land contamination	842	4	2612	5
Acidification	2095	2	2353	6
Eutrophication	557	6	1088	7

It is important to note that the benefit estimates presented for the various environmental issues offer only some guidance on environmental priorities for the Netherlands. In the absence of data on costs of implementing policies, these measures of effectiveness can provide only part of the picture necessary for establishing priorities. For a full scale economic analysis, like that in RIVM et al. (2000), benefit (damage) estimates need to be compared with cost estimates within a CBA framework. This is outside the scope of this study, however, a separate paper on the issues relating to and experience in such CBAs is prepared as Part II of the overall study (see *Part II: Integrating Cost-Benefit Analysis into the Policy Process*).

The caveats associated with the benefit estimates are mainly: i) not all benefits are estimated and ii) there is uncertainty about the ranges of values. Casual commentators argue that the existence of uncertainty undermines the credibility of the benefit estimates as a tool. It is our professional opinion that the presence of large uncertainty makes it more essential that benefit assessment is conducted. It serves to both increase the knowledge base in the area and reduce overall uncertainty and it acts as a signal to policy makers the potential risks of their actions. Although, it is interesting to see environmental issues ranked in terms of priority, the procedure is not sensitive to the uncertainty associated with the damage estimates for each environmental problem. To qualify the final rankings, the final column of *Table 4.8.2* presents an assessment of the degree of uncertainty, on a sliding scale of ++ to --, where ++ suggests low uncertainty, -- suggests very high uncertainty.

Although damage estimates are rising over time for the environmental issues (with the exception of eutrophication and acidification) as a proportion of Dutch GDP in 1995, 2010, 2020 and 2030 they fall. *Table 4.8.4* presents the damage estimates as a percent of GDP, (for details of Dutch GDP see *Annex 5 on Environmental Data, Assumptions and Models*). Note however that the falling percentage of GDP results makes no allowance for a rising relative value of the environment over time as income rises. If these relative valuations rise at the same rates as GDP, the proportion of damage to GDP would remain the same.

Table 4.8.4 Environmental damage estimates as a percent of Dutch GDP: %

	1995	*2010*	*2020*	*2030*
Low level ozone	1.32	1.12	1.01	1.02
PM10	0.76	0.53	0.47	0.46
Climate change	0.62	0.40	0.33	0.27
Acidification	0.85 - 0.95	0.30 - 0.34	0.23 - 0.25	0.18 - 0.20
Eutrophication	0.22 – 0.45	0.08 – 0.16	0.06 – 0.12	0.05 – 0.09
Noise	0.51	0.38	0.30	0.25
Total	4.29 - 4.62	2.82 - 2.93	2.40 - 2.48	2.22 - 2.29

Table 4.8.4 suggests that environmental damage in the Netherlands was a significant proportion of GDP in 1995, ranging from 1.3% for low level ozone, between 0.6% and0.8% for PM_{10}, climate change and acidification, to roughly 0.5% for noise and eutrophication. Overall, total environmental damage due to low level ozone, PM_{10}, climate change, acidification, eutrophication and noise is estimated to be roughly 4.5% GDP in 1995, falling to about 2% in 2030. It is interesting to compare these figures with the estimates of expenditure on pollution abatement, reported to be about 1.2% of GDP in 1990 (ERECO, 1992).

4.8.2 Burden of disease associated with selected environmental exposures

Introduction

To describe and compare the disease burden associated with environmental exposures, and, eventually, to perform cost effectiveness analysis of options for environmental policy. Some sort of 'public health currency' is required. Considering the fact that annual mortality or even loss of life expectancy does not fully represent the environmental health loss, we tentatively applied an approach largely based on the 'burden of disease' measure that was developed by Murray and Lopez on behalf of the World Bank and WHO (1995). To assess the global disease burden, and consequently the health policy priorities in different regions in the world, they employed 'disability adjusted life years' (DALYs). This health impact measure combines years of life lost and years lived with disease or disability that are standardised by means of severity weights, (World Bank, 1999 and Murray and Lopez, 1996). The notion that the multiform health loss due to environmental exposure is fairly well characterised by three dominant aspects of public health, viz. quantity of life (life expectancy), quality of life, and social magnitude or number of people affected inspired our adaptation of the DALY-concept. Provisional calculations for the Netherlands indicated that the contribution of environmental exposures to the total disease burden would probably not exceed 3%, which is roughly equivalent with the burden caused by car accidents, Hollander et al. (1999).

Figure 4.1 sketches the basic idea behind our approach. At birth potentially each of us may expect around eighty years of healthy life. However, our genetic program, our often-unfavourable life-styles, poverty, occupational or environmental conditions or just bad luck, means most of us will encounter disease that will reduce the quality of part of our life. These diseases may manifest themselves in episodes, chronic or even progressive until death. Some of us will die abruptly, for instance caused by an accident or an infectious disease. Thus, public health loss is defined as time spent with reduced quality of life, aggregated over the population involved. The methodology to estimate burden of disease associated with environmental exposures is described elsewhere (see Hollander et al., 1999).

Calculations

In the framework of the 5th National Environmental Outlook adequate data and future projections are only available for particulate matter, tropospheric ozone, noise, UV, radon, home dampness and food borne infectious disease. For each relevant health outcome we calculated attributable risks by combining population weighted exposure distributions with relative risk estimates derived from the epidemiological literature. Subsequently for each health outcome the number of cases was estimated by combining baseline incidence rates with the attributive risks. Calculations of future disease burden are based on projections of future population structure. *Table 4.8.5* presents the set of endpoints we used to arrive at estimates of attributable disease burden and the number of DALYs lost. Finally a total exposure attributable disease burden was calculated by aggregating the number of DALYs for each health outcome. The disease burden associated with additional UV-exposure due to ozone layer degradation was calculated by aggregating annual morbidity and mortality estimates of skin cancer and Dutch burden of disease data (Melse et al., 2000). Statistical uncertainty was assessed using MonteCarlo techniques.

Table 4.8.5 *Disability-Adjusted Life Years (DALYs) per 1000 cases lost to air pollution and UV-radiation.* *5-95%-tile (mean)*

Environmental factor	**Health outcome**	
Particulate air pollution	*Mortality*	*55-830*
Short-term	- respiratory	
	- coronary heart dis	
	- pneumonia	
	- other	
	hospital admission	
	- respiratory	11-34
	- cardiovascular	12-36
	emergency room visits	
	- respiratory	3.5-33.5
	aggravation of asthma	*0.1-1.0*
	- asthmatic attacks	
	- use of bronchodilators	
	aggravation of resp. symptoms	
	- upper respiratory. Tract	0-1.8
	- lower respiratory. Tract	1-15
	affected lung function	
	- decreased FEV1 >10%	0
Ozone	*Mortality*	*55-830*
	- respiratory	
	- cardiovascular	
	- pneumonia	
	- other	
	hospital admission	
	- respiratory disease	11-34
	emergency room visits	
	- respiratory disease	11-34
UV-A/UV-B	*Mortality*	*24,000 (mean)*
O_3-layer degradation	Melanoma	
	Other	
	Morbidity	
	Melanoma	650-1200
	basal cell	4-17
	squamous cell	58-190

Indoor radon	*Mortality*	
	lung cancer	13,500
	Morbidity	
	lung cancer	640-1150
Home dampness	*Morbidity*	
	Asthma	45-112
	chronic respiratory symptoms	10-74
Noise	*Mortality*	*11.000 (mean)*
	Cardiovascular	
	Morbidity	
	Hypertension	
	acute myocardial infarction	
	angina pectoris	200-350
	cerebrovascular disease	500-720
Food borne infectious disease	*Mortality*	*13,500 (mean)*
- Campylobacter spp.	Complications	
- Salmonella spp.	*Morbidity*	*1.9-3.8*
- SRSV	gastro-enteritis	
- E. coli (VTEC 0157)	Guillain-Barré syndrome	
- toxin producing	reactive arthritis	
(C. perfigens, B. cereus)	haemolytic uraemic syndrome (HUS)	

Findings
Figure 4.2 shows the annual disease burden for 2000, and 2030 (for the European Co-ordination scenario, in some cases 2020). The health effects of air pollution (particulates, and at some distance, troposheric ozone) dominate the disease burden associated with the set of environmental exposures. The future disease burden is to a large extent the result of future changes in the population structure (higher share of elderly people that are affected by this type of air pollution). Another environmental issue associated with a high disease burden is noise exposure (both from road and air traffic). We refrained from attributing disease burden to the large number of people reporting serious annoyance and sleep disturbance. There is much discussion whether these responses should be regarded as a damage to human health or rather as (merely) a social response. Instead we estimated the possible fraction of cardiovascular disease attributable to noise exposure based on the results of several large epidemiological studies implicating a causal association. Disease burden caused by chemicals (heavy metals, PAHs and benzene) is relatively minor (and is not presented in the figure). The contribution of UV radiation also appears to be relatively minor (however, attributive disease burden will rapidly increase until after 2050 due to a large time lag).

The figure also shows the estimates for indoor problems (radon and home dampness) and food borne infectious disease. Although the disease burden is high, it is questionable if these problems can be considered as environmental and are subject to environmental policies. However, the estimates can be used as reference points. The findings suggest that the disease burden associated with air pollution (particulate matter and tropospheric ozone) and noise exposure are substantial. Future policies focusing on these issues may yield a fair public health benefit for 2030.

Uncertainties and caveats
It has to be noted that the estimates for health effects of air pollution and noise are based on the results of epidemiological studies involving serious methodological problems (e.g. borderline resolution, confounding, external validity). Attributable disease burden numbers should be considered as maximum impact estimates.

It is important to note that attributable disease burden is only one of the many aspects that characterise environmental risk. Other aspects that determine the social acceptability of environmental risk are voluntariness of exposure, equity of the distribution of risks and benefits, trust in risk managers and government, and perceived controllability. Catastrophic potential is another important attribute: several studies have shown that accident that victimise a large number of people at the same time, such as aircraft or nuclear accidents, are much less accepted than accidents that kill small numbers of people a time, e.g. traffic accidents. The social disruption that takes place in response to large accidents may justify putting more weight on potential victims of nuclear accidents than on the faceless victims of air pollution or UV radiation.

Comparison with monetary benefit results: robustness test

The results of the damage assessment methodology presented in the earlier parts of *Chapter 4* and summarised in *Tables 4.8.1, 4.8.2 and 4.8.3* are in support of those derived from the DALY analysis. Both approaches suggest exposure to low level ozone and particulate matter are two of the top three priority environmental issues in the Netherlands. Noise scores least priority in terms of damage assessment. However, the results are not directly comparable with the DALY results because the damage assessment approach values noise in terms of WTP from CVM studies and house value and not specifically cardiovascular disease.

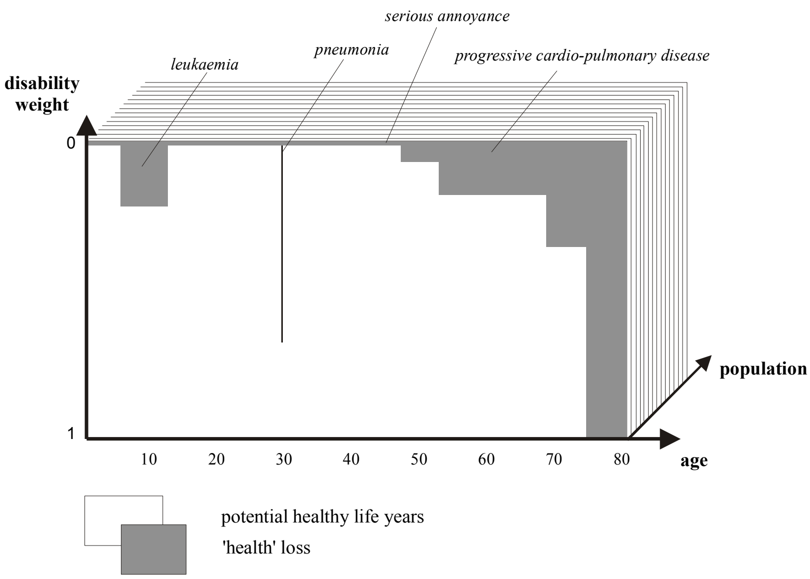
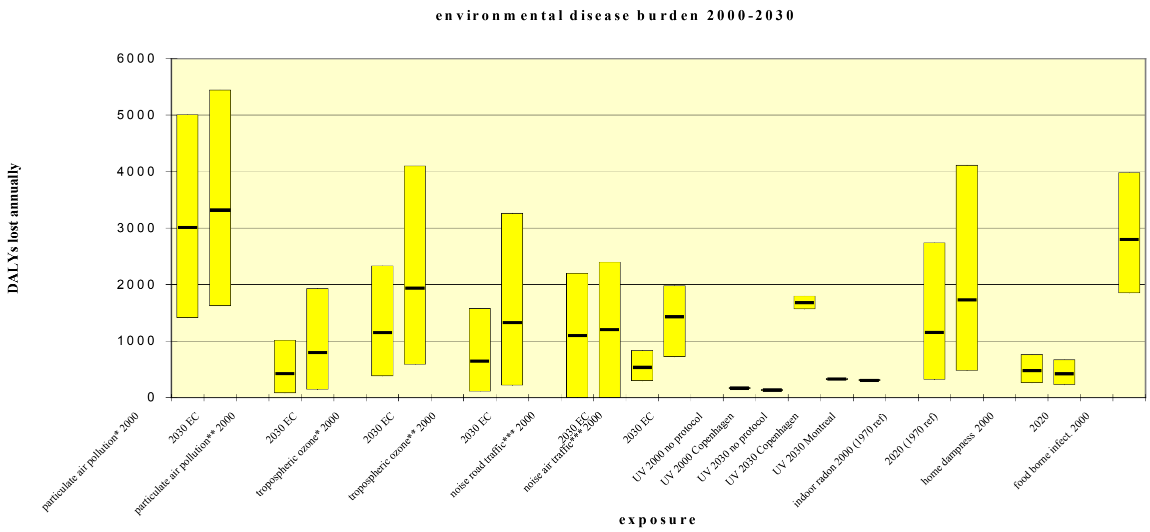


Figure 4.1. Diagram of the concept of disability adjusted life years



∗comprises all short term health outcomes; ** comprises only precipitated mortality and unscheduled hospital admissions; ***comprises only clinical health outcomes (no annoyance/sleep disturbance), must be regarded as 'maximal' impact, given

Figure 4.2. Disability-Adjusted Life Years (DALYs) lost annually to selected environmental exposures in 2000 and 2030; bars represent 5 and 95%-tiles of probability distribution.

5. Public opinion in the Netherlands

The number of Dutch people that consider environmental protection and the fight against pollution as an 'urgent and immediate problem' has fluctuated slightly throughout the last decade. Although the environment was seen as less of a problem in 1999 than in 1995, when the concern was at its highest (63% in 1986, 80% in 1995 and 70% in 1999) *(*Eurobarometer, 1986, 1995*),* it remains a common concern for the Dutch.

5.1 Environmental issues in general

When considering environmental issues, there is a distinction between threats to the environment, threats to family health, local problems and national / international issues. All issues cited in this summary are seen as being more threatening to the environment rather than health.

Drawing upon all the survey results, the issues considered to constitute a 'serious threat to the environment' (regardless of locality) are listed in *Table 5.1*.

Table 5.1 Issues considered to constitute serious environmental damage

Issue	Ranking
Factories releasing dangerous chemicals into the air or water	1
Oil pollution of the seas and coasts	2
Global pollution (gradual disappearance of tropical forests, destruction of the ozone layer, greenhouse effect etc)	3
Storage of nuclear waste	4
Industrial Waste	5
Acid Rain	6
Excessive use of herbicides, insecticides and fertilisers in agriculture	7
Rubbish in the streets, in green spaces or on beaches	8
Air pollution from cars	9
Uncontrolled mass tourism	10
Sewage	11
Noise generated by building or public works, heavy traffic, airports	12

Source: Eurobarometer: Europeans and their Environment, 1992, 1995 and SCP survey 1993.

These opinions are fairly resilient to time and results from 1992 show similar rankings, (excepting the issue of acid rain, which has increased in importance slightly since 1992).

5.2 Global environmental issues

Global concerns are still high on the public agenda as listed in *Table 5.2* and there is no sign of 'environmental fatigue'. When asked about global threats to the environment, the destruction of tropical forests was seen as 'very worrying' and as the most immediate threat. This was followed by the disappearance of biodiversity and threats to natural habitats and the risk that pollution from the industrialised countries spreads to less industrialised countries. The public are also 'strongly concerned' about the depletion of natural resources.

Seen as less of a threat, but, still 'worrying' to the public are the issues of global warming, (which was seen as a threat to both environment and personal health), acid rain and the depletion of the ozone layer. Despite the coverage of these issues in the media over the last decade, there is little evidence of public 'fatigue'.

Table 5.2 International environmental concerns

Issue	*Average score (4-very worried, 1 – not at all worried)*	*Ranking*
Disappearance of tropical forests	3.6	1
Disappearance of certain types of plants, animals and habitats throughout the world	3.3	2
Risk that pollution from the industrialised countries spreads to less industrialised countries	3.3	2
Depletion of natural resources throughout the world	3.2	3
Global warming (greenhouse effect)	3.1	4
Destruction of the ozone layer	3.1	4
Acid Rain	3.1	4

Source: Eurobarometer, 1992 and 1995.

5.3 National environmental issues

There is a slight difference in this prioritisation when considering threats within 'one's own country'. In this case, the results are listed below in *Table 5.3*.

Table 5.3 Concerns when considering one's own country

Issue	*Average Score 4-very worried, 1-not very worried*	*Ranking*
Industrial waste	3.1	1
Pollution of seas and coasts	3.0	2
Harm caused to animals, plants and the natural habitat	3.0	2
Risks connected with the use of nuclear power	3.0	2
Air pollution	2.9	3
Pollution of rivers and lakes	2.9	3
Pollution of agricultural origin (insecticides, slurry etc)	2.8	4
Urban sprawl	2.8	4
Possible risks for the environment from the development of new bio-technologies	2.7	5
Motor sports in the countryside like outboard, motor-cross, trials, off road, jet skis or snow skis	2.7	5
Risks connected with industrial activities	2.7	5
Natural disasters like floods, landslides, earthquakes, fires	2.5	6
Growth caused by tourism	2.5	6
Hunting	2.2	7

Source: Eurobarometer: Europeans and their Environment, 1992, 1995 and SCP survey 1993.

All the issues listed in *Table 5.3* provoke concern, with the scale between 'slightly worried' for hunting to 'worried' for issues such as industrial waste. The aspects of the environment provoking most concern are industrial waste, biodiversity loss and harm to animals, plants and the natural habitat, pollution of seas and coasts and pollution of rivers and lakes.

5.4 Attitudes towards the future

That environment is seen as an important issue does not stop the public being fairly confident about the future and the level of concern about environmental disaster is low among the public as is the level of comprehension of risk, as shown in *Table 5.4*. However, there is no sign of 'environmental fatigue' and support for more information campaigns on how to support the environment is very high as is the willingness to sacrifice personal activities for the sake of a better environment.

Table 5.4 Attitudes toward the future

Issue	*1=high, 6=low*	*Level of concern*
Support for info campaigns	1	high
Collective action	1	high
Support for pricing instruments	3	Middle
Conscious purchasing	4	Fairly low
Comprehension of risk from cars	6	low
Comprehension of risk	6	low
Concern for the future	6	low

Source: ISSP 1993 and Compass 1993.

When asked where the public obtain their environmental information, opinion shows a healthy trust in environmental institutions but this trust has been slightly shifted between 1992-1995 from environmental institutions to scientists (SCP survey, 1997).

5.5 Environmental protection action- who is responsible?

Government policy is still seen as a necessary condition for the solution of diverse environmental problems, and influencing politicians and government policy is still the key aspect of the activities of social organisations. Also indicated is that environmental organisations are in a transition period in their relationship with the Government. But, still, the Government is seen as having primary responsibility – that there are a great many environmental issues which are too large for the environmental organisations and their networks to handle. When the question of 'how to improve the environment' is addressed, the public are reasonably supportive of the polluter pays principle. However, in Holland, there is a preference for coercive measures and information campaigns and a very strong wish to have common environmental laws throughout Europe.

As far as public attitude and actions towards the environment go, although air pollution is regarded as possibly the most important threat, the public are reluctant to cite private cars as a major source of this pollution. Threats to the environment are seen to be more a result of bad industrial activity than personal activity, and although the majority of Dutch public support the introduction of green taxes, they are lagging in their personal application of measures to improve the environment and are average or well below the European average doing so.

6. Prioritisation of environmental issues

In order to determine a final ranking for the environmental issues in the Netherlands in order of importance we draw upon the results of the damage assessment (see *Chapter 4*), the public opinion in the Netherlands (see *Chapter 5*) and the DALY assessment (see *Section 4.8.2*). *Table 6.1* brings together the results of the three methods.

The final ranking for the environmental issues given in the end column of *Table 6.1* is based on a simple 'Borda count', i.e. for each environmental problem sum the weighted ranking from the public opinion, the damage assessment and the DALY procedure and divide by the number of total environmental issues considered. It is necessary to weight the rankings in order to allow for the different numbers of environmental issues considered in the different approaches, i.e. twelve issues are considered in the public opinion survey, seven in the damage assessment and seven in the DALY approach. For example, for climate change the 'Borda count' is found by the following calculation (5*7 + 3*12 = 71)/19. The overall ranking is found by ordering the results of the 'Borda count', where lower values score higher priorities. Although the 'Borda count' is a conventional way to rank a number of rankings, the main disadvantage of this procedure is that the overall rankings are not sensitive to the uncertainty associated with the damage assessment for each environmental problem. To qualify the final rankings, *Table 6.1* includes an assessment of overall uncertainty for each problem in the final column, using a scale of '++ to –-', where ++ indicates low uncertainty and -- indicates high uncertainty.

Table 6.1 Environmental issues in the Netherlands in order of priority

Environmental problem	Ranking according to damage assessment	Ranking according to public opinion	ranking according to DALYs	'Borda count'	final ranking	uncertainty
Land contamination	2	-	-	14/7	1	--
Climate change	5	3	-	71/19	2	++
PM10	3	9	1	136/26	3	+
Acidification	4	6	-	100/19	4	++
Low level ozone	1	9	4	143/26	5	+
Eutrophication	7	7	-	133/19	6	--
Noise	6	12	3	207/26	7	-
No of issues considered in survey	*7*	*12*	*7*			

Note that, public opinion rankings are taken from the results of the Eurobarometer (1992, 1995). These surveys ask the Dutch public their opinion about environmental issues, regardless of locality, and most of the environmental issues analysed in this study are included. However, it is necessary to make some assumptions, firstly, we assume that 'excessive use of herbicides, insecticides and fertilisers in agriculture' corresponds to 'eutrophication' as defined in this study, although we acknowledge this is an imprecise correspondence as there are other impacts of 'excessive' use of these chemicals, whilst 'air pollution from cars' is assumed to represent both PM_{10} and low level ozone. Unfortunately land contamination was not included in the public opinion surveys. The rankings for public opinion will not necessarily be the same as the damage assessment results. There should be some links because if people say they care then generally this means they are willing to pay to prevent environmental damage.

Although the damage assessment is considered very uncertain for land contamination, we include it in the overall ranking of environmental issue in order to determine whether the Netherlands is right to commit so much expenditure to this issue.

From *Table 6.1* we can conclude that:

- land contamination scores 2nd priority from the damage assessment, whilst the overall ranking method places it at top priority. As *Table 6.1* indicates the damage assessment is considered to be very uncertain;

- climate change is ranked 3rd out of twelve problems considered by the public opinion survey, if we rank the public opinion results according to the seven issues considered in this study, climate change would be the top priority. The damage assessment procedure gives climate change a high ranking at 5th place. The overall 'Borda count' ranking suggests climate change is one of the top priority issues for the Netherlands at 2nd place;

- PM10 is ranked 3rd by the damage assessment and 3rd by the Dutch public when ranked according to the seven issues of this study only and 1st by the DALY approach. Overall particulate matter is one of the top three environmental issues;

- the Dutch public rank acidification 6th out of twelve problems or 2nd priority when ranked according to the seven issues of this study only, whilst the damage assessment methodology gives acidification a priority at 4th. The overall ranking, based on the 'Borda count' suggests acidification is 4th;

- low level ozone is ranked 9th out of twelve problems by Dutch public opinion, which corresponds to 3rd, if we rank according to the seven issues of this study only. The damage assessment results rank this environmental problem as a top priority and the DALY methodology suggests low level ozone ranks 4th out of seven issues. Overall the 'Borda count' result suggests low level ozone is not a priority issue for the Netherlands and places low level ozone at 5th;

- eutrophication is given a low priority by both the Dutch public (7 out of twelve problems, or 5th out of seven problems) and the damage assessment procedure at 7th. This is reflected in the overall ranking place of second least priority. Note, however that the damage estimates for eutrophication are acknowledged as extremely uncertain;

- noise is ranked as the lowest priority problem by the Dutch public. The damage assessment methodology also ranks noise low at 6th, whilst the DALY approach ranks noise as 3rd highest priority out of seven issues. The final ranking suggests noise nuisance is least priority issue for the Netherlands.

Definitions

2xCO_2: the scientific research on global warming impacts has almost entirely focused on the case of CO_2 concentration doubling, i.e. the impacts of an atmospheric CO_2 concentration of twice the preindustrial level (2 x CO_2). This a completely arbitrary benchmark, chosen soley for analytical convenience. It is neither an optimal point nor a steady state and warming will continue and may in fact worsen beyond 2 x CO_2.

Annuitisation factors: sometimes it is more useful to represent damage estimates (given as present values using an agreed discount factor) as an annual sum, i.e. an annuity. An annuity is simply an annual sum, which when discounted at the relevant discount rate, would give the present value in question. We use the following formula to establish the relevant annuity, A:
A = PV. (s/(1+s))
Where A = annuity, PV = present value, s= discount rate

Contingent valuation methodology: estimates the WTP for a change in the quantity and/or quality of an environmental good by using survey techniques (Mitchell and Carson, 1989). WTP is estimated by way of implementing a structured questionnaire to a representative sample of the population of concern. In the questionnaire a hypothetical change is described and the respondent is asked directly for her WTP for this change. The valuation questions are supplemented by questions on socio-economic characteristics and relevant attitudes and preferences regarding the good in question. This information is used to estimate a valuation function which 'explains' WTP as a function of these variables. In order to obtain a valid response it is crucial to provide an accurate and meaningful description of: (i) the change that is valued, (ii) the way and how often the respondent is expected to pay and also respondents must be reminded of their budget constraints. For further details see Freeman (1993).

Cost - benefit analysis: procedure for estimating gains (benefits) and losses (costs) of a project or policy in monetary terms and comparing these estimates.

Discount rate: the rate at which future costs and benefits are adjusted downwards to reflect, for example, that current benefits or costs are valued more highly than future costs or benefits.

Hedonic price technique: the underlying concept of the hedonic price technique is that the price of a good, for example, a house, is a function of its attributes, such as the number of rooms, size, location, and environmental characteristics, such as noise levels and ambient air quality and others. The HPT proceeds by estimating an hedonic price function by regressing house price on the relevant characteristics. In the simplest form of the method, a measure of the value of an environmental characteristic of interest, such as noise, can be deduced by differentiating the hedonic price function with respect to noise. For further details refer to Freeman (1993).

Purchasing Power Parity (PPP) is defined as the rates of currency conversion that equalises the purchasing power of different currencies by eliminating the differences in price levels between countries. They are relative prices. For example, if the price of cauliflower in France is 8.00 francs and in the United State it is 1.50 dollars, then the PPP for cauliflower between France and the US is 8.00francs to 1.50 dollars or 5.33francs to the dollar. This means that for every dollar spent on cauliflower in the United States, 5.33 francs would have to be spent in France to obtain the same quantity and quality or, in other words, the same volume of cauliflower, www.oecd.org/std/ppp/pppbackground.

Present value: is the value of future amounts in current terms the following formula is used to establish present value damage estimates over the period infinity:

$PV = X \cdot (1+s) / (s-g)$
Where PV = present value, X = undiscounted damages in 1995, s = discount rate (i.e. 6%) and g is the growth rate of damages over time, i.e. between 1995 and 2030.

Primary benefits: avoided environmental damage from issue A due to environmental control policies targeted at issue A.

Secondary benefits: avoided environmental damage from issue B due to environmental control policies targeted at issue A.

Total benefits: the sum of primary benefits and secondary benefits.

Uncertainty: deficiency in knowledge about events with an unknown probability of occurrence.

Value of a Statistical Life: the sum of individuals' own valuations of risks to their own lives (refer to *Annex A3*).

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Annex I Methodology and assumptions

1. Monetary valuation techniques

Introduction

The economic approach to valuing environmental changes is based on people's preferences for changes in the state of their environment. Environmental resources typically provide goods and services for which there are either no apparent markets or very imperfect markets, but which nevertheless can be important influences on people's well-being. Examples include the quality of air, which affects people's health, crop yields, damage to buildings, and acidification of forests and fresh waters.

However, the lack of markets for some of these services and existence of imperfect markets for others means that unlike man-made products, they are not priced, therefore their monetary values to people cannot be readily observed. The underlying principle for economic valuation of environmental resources, just as for man-made products, is that people's *willingness to pay* (WTP) for an environmental benefit, or conversely, their *willingness to accept compensation* (WTA) for environmental degradation, is the appropriate basis for valuation.

If these quantities can be measured, then economic valuation allows environmental impacts to be compared on the same basis as financial costs and benefits of the different scenarios for environmental pollution control. This then permits an evaluation of the net social costs and benefits of each scenario for each environmental issue.

The lack of markets and prices for many environmental goods and services means that the challenge for economists is twofold. The first task is to *identify* the ways in which an environmental change affects well-being. This is addressed in the next section, where the components of 'total economic value' of a resource are explained. The second task is to *estimate the value of these changes* through a variety of direct and indirect valuation techniques, exposition of which is given in the following sections.

Total Economic Value

The monetary measure of the change in society's well-being due to a change in the quantity and / or quality of environmental assets or quality is called the total economic value (TEV) of the change. To account for the fact that a given environmental resource provides a variety of services to society, TEV can be disaggregated to consider the effects of changes on all aspects of well-being influenced by the existence of the resource.

TEV can be divided into *use values* and *non-use values*, the latter also being called 'passive use values'. Use values include:

- direct use values, where individuals make actual use of a resource for either commercial purposes (e.g. harvesting timber from a forest) or recreation (e.g. swimming in a lake);
- indirect use values, where society benefits from ecosystem functions (e.g. watershed protection or carbon sequestration by forests), and
- option values, where individuals are willing to pay for the option of using a resource in the future (for example, future visits to a wilderness area).

Non-use values can take the form of:

- existence values, which reflect the fact that people value resources for 'moral' reasons, unrelated to current or future use
- bequest values, which measure people's willingness to pay to ensure their heirs will be able to use a resource in the future

Typically it is not always easy or even necessary to separate existence and bequest values.

To arrive at an estimate of the net change in societal well-being arising from an environmental change, we must consider each of these elements in turn. The total economic value (TEV) of a change is the sum of both use and non-use values:

TEV = use values + non-use values

= direct use + indirect use + option + existence + bequest values

Table 1.1 presents a taxonomy for environmental resource valuation, using the total economic value of a forest as an illustration.

Table 1.1 Economic taxonomy for environmental resource valuation

Total Economic Value				
Use Values			Non-use Values	
Direct Use	Indirect Use	Option Value	Bequest Value	Existence Value
Outputs directly consumable	functional benefits	Future direct and indirect values	use and non-use value of environmental legacy	value from knowledge of continued existence
• food • biomass • recreation • health	• flood control • storm protection • nutrient cycles	• biodiversity • conserved habitats	• habitats • prevention of irreversible change	• habitats • species • genetic • ecosystem

The first step in estimating any of these values is the definition and measurement of the environmental impact. This often includes an element of scientific uncertainty that can, at times, be quite significant. The accuracy of economic valuation is therefore dependent on accurate scientific identification and quantification of the environmental change in order to estimate people's preferences for or against it.

Valuation Techniques

The practical problem with economic valuation is one of deriving credible estimates of people's preferences in contexts where there are either no apparent markets, or very imperfect markets. In the case of marketed goods, the market price is the measure of willingness to pay and can be readily observed. However, in the case of non-marketed goods and services we need to elicit this value in different ways. There are two broad approaches to valuation, each comprising several different techniques, as illustrated in *Figure 1.1*.

- **Revealed preference techniques,** which infer preferences from actual, observed market-based information. Preferences for environmental goods are revealed indirectly when individuals purchase marketed goods which are related to the environmental good in some way, and
- **Stated preference techniques,** which attempt to elicit preferences directly by use of questionnaire. All valuation of non-use values depends on these techniques.

We consider each of these approaches in turn, highlighting when each could be used, their advantages and drawbacks .

Revealed Preference Techniques

The essence of revealed preference techniques is that they infer environmental values from markets in which environmental factors have an influence. For example, there are markets for certain goods to which environmental commodities are related, as either substitutes or complements to the goods in question. In this way people's actions in actual markets reflect, to a certain extent, their preferences for environmental assets.

There are four main revealed preference techniques that are considered in the sections that follow.

1. Averting behaviour
2. Hedonic pricing (of property and labour)
3. Travel cost method
4. Random utility and discrete choice modelling

Averting Behaviour

The basis for the averting behaviour technique is the observation that marketed goods can act as substitutes for environmental goods in certain circumstances. When a decline in environmental quality occurs, expenditures can be made to mitigate the effects and protect the household from welfare reductions. For instance, expenditure on sound insulation can indicate households' valuation of noise reduction; expenditure on household water filters can be used to estimate economic values of clean water.

The method is applicable in situations where households spend money to offset environmental impacts. It requires data on the environmental change and its associated substitution effects. Fairly crude approximations can be found by simply looking directly at changes in expenditures on the substitute good resulting from some environmental change.

Advantages of these models are that they have relatively modest data requirements and can provide theoretically sound estimates based on actual expenditures. However, they can give incorrect estimates if other important aspects of individuals' behavioural responses are ignored. For example, individuals may engage in more than one form of averting behaviour in response to any one environmental change. Additionally, the averting behaviour may have other beneficial effects that are not considered explicitly, for example sound insulation may also reduce heat loss from a home. Furthermore, averting behaviour is often not a continuous decision but a discrete one: for example, a smoke alarm is either purchased or not. In this case the technique will tend to underestimate the value of the environmental good.

Hedonic Pricing

This technique involves the analysis of existing markets where environmental factors have an influence on price. The example most frequently used is that of the housing market, as the environmental attributes of the surrounding area have an impact on the price of property. For example, noise levels will be higher close to an airport and, other characteristics being equal, this can be expected to lower the price of a property in that area. Similarly, two identical properties which differ only in, say, the local air quality, will differ in value to the extent that people find one air quality preferable to the other. The difference can be viewed as the value attached to the difference in air quality as measured by WTP.

The hedonic property price method can be used even when properties differ in many factors other than environmental quality provided that data are detailed enough. With the use of appropriate statistical techniques, the hedonic approach attempts to (i) identify how much of a price differential is due to a particular environmental difference between properties, and (ii) infer how much people are willing to pay for an improvement in environmental quality that they face and what the social value of the improvement is. The main disadvantage of the hedonic property price method is that the analysis is very data intensive and it requires a well functioning property market.

The same technique has also been applied to the labour market in the valuation of work-related risk in hedonic wage (HW) studies. Identification of wage differentials due to differences in health risks, for example, will give an indication of willingness to accept compensation (WTA) for incurring these risks, which can be used as a measure of the benefits of improving safety.

Travel Cost Method

Many natural resources are used extensively for the purpose of recreation. It is often difficult, however, to value these resources because no prices generally exist for them. The travel cost approach is based on the fact that, in many cases, a trip to a recreational site requires an individual to incur costs in terms of travel, entry fees, on-site expenditures and time. These costs of consuming the services of the environmental asset are used as a proxy for the value of the recreation site and changes in its quality.

Clearly, because travel cost models are concerned with active participation they measure only the use value associated with any recreation site. The method is now well-established as a technique for valuing the non-market benefits of outdoor recreation resources. It is useful because it is based on actual observed behaviour. However, the technical and data requirements are such that it is not readily applicable.

Random Utility or Discrete Choice Models

While the travel cost method is useful for measuring total demand or WTP for a recreational site, this technique is less useful for estimating the value of particular features or assets of the site which may be of interest. Random utility models have been developed for this purpose.

The emphasis of random utility or ‘discrete choice’ models is on explaining the choice between two or more goods with varying environmental attributes as a function of their characteristics. This can be useful where, for example, polluting activity causes damage to some features of a recreational site but leaves others relatively unharmed.

This can be illustrated using a simple example from a choice of transport mode. Supposing that, when undertaking a given journey, an individual faces the choice of travelling by taxi or by public transport. A taxi will take 20 minutes and cost Euro5, whereas public transport will take an hour but cost Euro 2. If the individual chooses to travel by taxi, it can be inferred that s/he judges the difference of 40 minutes in time to be worth at least the Euro 3 difference in fare. In other words, the value of the individual’s time is at least Euro4.50 per hour.

Another example is the choice between bottled water and tap water for drinking. The former is more expensive but associated with better quality. Therefore, the price difference between bottled and tap water is an indication of the value of risk in this context.

Stated Preference Techniques

Stated preference techniques enable economic values to be estimated for a wide range of commodities which are not traded in markets. In addition, these techniques are the only way to estimate non-use value of environmental resources. Here, we consider two approaches:

1. Contingent valuation, and
2. Choice modelling.

Contingent Valuation Method

In contingent valuation method (CVM) studies, people are asked directly to state what they are willing to pay for a benefit or to avoid a cost, or, conversely, what they are willing to accept to forego a benefit or tolerate a cost. A contingent market defines the good itself, the institutional context in which it would be provided, and the way it would be financed. The situation the respondent is asked to value is hypothetical (hence, 'contingent') although respondents are assumed to behave as though they were in a real market. Structured questions and various forms of 'bidding game' can be devised to assess the maximum willingness to pay. Econometric techniques are then applied to the survey results to derive the average bid value, i.e. the average WTP.

There are three basic parts to most CVM surveys. First, a hypothetical description of the terms under which the good or service is to be offered is presented to the respondent. Information is provided on the quality and reliability of provision, timing and logistics, and the method of payment. Second, the respondent is asked questions to determine how much s/he would value a good or service if confronted with the opportunity to obtain it under the specified terms and conditions. These questions take the form of asking how much an individual is willing to pay for some change in provision. Respondents are reminded of the need to make compensating adjustments in other types of expenditure to accommodate this additional financial transaction. Econometric models are then used to infer WTP for or WTA the change. Finally, questions about the socio-economic and demographic characteristics of the respondent are asked in order to relate the answers respondents give to the valuation question to other characteristics of the respondent, and to those of the policy-relevant population.

Given that the accuracy of results also largely depends on careful construction of the survey, a set of guidelines for applying CV to derive reliable estimates of non-use values is developed by the US National Oceanic and Atmospheric Administration (NOAA) panel (Arrow et al, 1993). An updated and extended version of these guidelines is prepared by EFTEC for the UK Department of Environment, Transport and the Regions (EFTEC, 2000).

Choice modelling

Choice modelling (CM) is a broad term used to cover several different techniques, all of which are survey methods, involving asking individuals to rank alternatives rather than explicitly express a WTP or WTA. The inclusion of prices as one of the attributes of each alternatives enables monetary values to be derived from the rankings. Other aspects are similar to contingent valuation. The main applications have been in the context of human health and landscape effects, as well as disamenity.

Dose- and Exposure-Response Functions

Dose response functions (DRFs) measure the relationship between a unit concentration of a pollutant and its impact on the relevant receptor. Exposure-response functions (ERFs) are based on the same principle but measure the response with respect to the exposure. Exposure is a measure of the levels of a pollutant in the environment surrounding the receptor in question. For example, a person may be exposed to a certain concentration of an atmospheric pollutant, but the dose received will depend on the amount inhaled, which is higher during exercise and lower during rest. In general, effects will be more closely related to dose, but it is much easier to measure exposure. Hence it is important to recognise that any dose-response function is often represented by the approximation of an exposure-response function (ApSimon et al., 1997).

Dose-response functions are used extensively where a physical relationship between some cause of damage, such as pollution, and an environmental impact or 'response' is known and can be measured. Once the relationship has been estimated, then WTP measures derived from either conventional market prices (which are adjusted if markets are not efficient) or revealed / stated prices (where no markets exist) using one of the techniques described in the above section. The physical damage is multiplied by this WTP, or value per unit of physical damage, to give a 'monetary damage function'.

The approach is theoretically sound, and can be used wherever the physical and ecological relationships between a pollutant and its output or impact are known. The specification of the D/ERF is crucial to the accuracy of monetary valuation and is usually the main source of uncertainty. Difficulties and uncertainties may arise in: identifying the pollutant responsible for the damage and all possible variables affected; isolating the effects of different causes to determine the impact on a receptor, e.g. synergistic effects where several pollutants or sources exist; identification of damage threshold levels and the long term effects of low to medium levels of pollution. All these problems make it difficult to determine the appropriate empirical specification of the functional form. Additionally, there is the further complication that evidence of a physical response may not be economically relevant if individuals are not concerned about it and, therefore, do not attach a value to avoiding it. For these reasons, large quantities of data may be required and the approach may be costly to undertake.

If, however, the D/ERFs already exist and the impacts are marginal, borrowing the functions is very inexpensive and provide reasonable first approximations to the true economic value measures.

2. Benefits transfer

Benefits transfer (BT) is the basis of both the 'bottom-up' and 'top-down' approaches discussed in this study. BT makes use of previous valuations of similar environmental changes in other regions and countries, and, with any necessary adjustments, applies them to produce estimates for the specific environmental damage involved in the present study. Thus, it relies on both methodology and data from previous studies in transferring the results to derive estimates of the damage due to the environmental issues under consideration.

The issue of relying on actual data from a previous study is fraught with difficulty. In the BT approach, data on physical impacts, geographical aspects and local populations from a previous study are transferred and applied to the Netherlands. This procedure is most accurate when impacts are not affected by local characteristics. Thus, for global impacts (such as the impacts due to global warming from GHG emissions) the use of BT is easily justified. However, when impacts are local, much greater care is required in transferring results of previous valuations in other regions or countries. If circumstances differ between the original study site and the site in the Netherlands, accuracy is likely to be considerably reduced. Adjustments can be made for:

- average income;
- population size and characteristics;
- background conditions;
- level of impacts, and
- other determinants for which there are accessible data for the current study.

But there remains the problem that different populations may hold very different attitudes towards environmental concerns, for cultural, educational, or other reasons. Thus significant differences in WTP of different communities for similar environmental impacts may remain even after these adjustments.

Boyle and Bergstrom (1992) propose that successful BT requires

- similarity of the environmental good to be valued;

- similar population characteristics, such as income, age structure and education in the compared sites, and

- evidence of sound economic and statistical techniques used in the original studies.

The process of BT is clearly less than ideal. BT is only as accurate as the original studies. Thus, the quality of the data and the methodology of the original studies need to be examined.

The attractions of benefits transfer are very clear: without it, it would be necessary to resort to original valuation studies. This is both expensive and time consuming, which is a problem for a wide-ranging study such as this one, where we require valuations across many environmental issues.

Application of benefit transfer in this study

In this study all values are reported in Euro as valued in January 2000. Translation of WTP values into Euro is more complicated than the simple use of financial exchange rates. In general, environmental goods can be considered as a consumable, i.e. clean air, improved health, etc, this means it is something that respondents would 'buy' with their disposable income in order to generate utility or welfare. The decision of where to 'buy' the environmental good at the price given is therefore critically dependent on the prevailing prices at which other consumable goods can be purchased. For many reasons, similar market goods cost different amounts of money in different countries. It is these price differences that must be considered when converting WTP values from one currency to another.

Ready et al. (1999) state that the correct exchange rate to use when converting WTP values is the exchange rate that holds purchasing power constant, rather than the exchange rates seen in the financial markets.

OECD publishes a set of purchasing power parity exchange rates (PPPs) (available on their website), that reflect differences in the national average prices for the standardised bundle of goods. Where environmental goods are broadly distributed nationally, the nation-wide estimates of the PPP-adjusted exchange rates provided by OECD are appropriate for this task. On the other hand where impacts are concentrated in a region with higher prices (such as a city), then a more region-specific measure may be needed. For example, if costs of a representative basket of goods and services in the target city are 10% above the national average, an appropriate estimate of the city-specific rate would be 1.1 times the nation-wide rate.

All WTP values estimated for any country other than the Netherlands are converted to WTP values specifically for the Netherlands and all WTP values are given as Euro at 2000 prices. This is achieved by the following procedure:

Adjustment 1: Spatial adjustment

Spatial adjustment describes the procedure of adjusting WTP values for the UK,for instance, to estimate Dutch WTP values. This is achieved by using Purchasing Power Parity adjusted exchange rates provided by OECD at:
www.oecd.org/std/ppp/pppbackground.

1. $\boldsymbol{WTP_{SOURCE\,.\,NATIONAL\ CURRENCY\,.\,YEAR\ A} \rightarrow WTP_{DUTCH\,.\,DUTCH\ GUILDERS.\ YEAR\ A}}$

From: $WTP_{DUTCH.\ GUILDERS.YEAR\ A} = WTP_{UK.\ STERLING.YEAR\ A} \times (PPP_D / PPP_{UK})$

Financial Exchange rates are used, in the relevant year, to convert Dutch guilders to Euro.

2. $\boldsymbol{WTP_{DUTCH.\ DUTCH\ GUILDERS.\ YEAR\ A} \rightarrow WTP_{DUTCH.EURO.YEAR\ A}}$

From: $WTP_{DUTCH.\ EURO.YEAR\ A} = WTP_{DUTCH.\ GUILDERS.YEAR\ A} \times$ (Euro year A / Guilder year A)

Adjustment 2: Intertemporal adjustment

Most WTP estimates are given in currencies that relate to previous years, such as 1990, 1991, 1992, etc. For the purposes of this study, all values are converted to Euro 2000 values by means of the *Harmonised Consumer Price Index* (HCPI), given in *table 2.1*. The corresponding deflators are reported in *table 2.2*.

3. $\boldsymbol{WTP_{TARGET.EURO.YEAR\ A} \rightarrow WTP_{TARGET.EURO.YEAR\ 2000}}$

From: $WTP_{DUTCH.EURO.YEAR\ 2000} = WTP_{DUTCH.EURO.YEAR\ A} \times$ (Euro year 2000 / Euro year A)

Table 2.1 *Harmonised consumer price index*

1985	1986	1987	1988	1989	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999	2000
100	-	106.9	110.7	116.3	121.5	-	-	134.3	144	148.5	152.2	154.8	157.2	159.7	162.3

RIVM (1999) for the years 1990 - 1997. Eurostat yearbook Data for 1987-1997: for the years 1997 -1998. Authors assume 1.6% increase p.a.: for the years 1998 - 2000.

Based on the above CPI the relevant deflators for adjusting values to Euro 2000 prices are as follows:

Table 2.2 *CPI deflators*

Year	Euro 2000
1987	1.6225
1988	1.4656
1989	1.3950
1990	1.3353
1991	n.k.
1992	n.k.
1993	1.2081
1994	1.1267
1995	1.0925
1996	1.0660
1997	1.0481
1998	1.0319
1999	1.0159

Source: RIVM (1999): for years 1986 - 1997 Eurostat Yearbook 1987 - 1997: for years 1997 - 1998, Authors own calculations: for the years 1998 - 2000.

3. Valuing the risk of premature mortality

The monetary estimates of the damage to human health from various pollutants and accidents tend to dominate the total damage estimates. Therefore, it is useful to have a special section to make clear the different concepts and methods for valuing the risk to human health.

For a change in risk that threatens life and health generally, the relevant valuation is the value that the individual at risk attaches to their own health and life chances, plus what others would be willing to pay to avoid the risk to that individual, plus any costs that society at large bears and which would not otherwise occur if the individual did not suffer the effects of the risk in question. The components of this value of risk (VOR) are:

(a) $VOR_{i,i}$ where 'VOR' refers to the individual i's valuation of risk to themselves, i.e. 'own risk'. The way in which these individual VORs are aggregated is dealt with shortly. Essentially, we will require the summation of such own valuations for all individuals at risk to give $\Sigma_i VOR_{i,i}$, more commonly known as the 'value of a statistical life'

(b) $VOR_{i,j}$ where the i,j notation now refers to j's valuation of risks to i. Again, this will need to be summed for all j, i.e. for all people expressing some concern about risks to i, to give $\Sigma_j VOR_{i,j}$

(c) COI_i where COI refers to the 'cost of illness' suffered by i but which costs are borne by the rest of society. An example would be hospital costs. COI could be regarded as part of VORi,j

The relevant literature focuses on (a) with very limited evidence on (b). If the rule that individuals' preferences expressed as willingness to pay or accept is followed, then the component (c) should not be included at the value of risk but expressed separately where relevant. Since the majority of values used in the literature as reported in this study are based on (a), the rest of this methodological section focuses on this.

Valuing Statistical Lives

The way a VOSL is obtained is by aggregating up from a WTP value of risk reduction. Imagine the probability of dying next year is 0.004 for each person and suppose we have 1000 persons in the population. Assume there is some risk reduction policy that reduces the risk to 0.003, a change of 0.001. Each person is asked to express their WTP for this change in risk and suppose the answer is Euro1000. The risk reduction policy is a public good: it affects everyone equally. Thus 1000 people say they are each willing to pay Euro1000 for the policy, i.e. their aggregate willingness to pay is Euro1 million. The change in risk will result in one statistical person being saved each year (1000 x 0.001). Thus the value of a statistical life is Euro1 million in this example. It is important to understand that no-one is being asked their WTP to avoid themselves dying at a specified time: they are being asked to express a WTP for a change in risk.

Individuals' WTP to reduce risks can be expected to vary across different individuals. The two main reasons for this will be that:

1. people have differing attitudes to risk: some may even be 'risk lovers', i.e. positively enjoying risky contexts. Most people are risk avoiders, i.e. they will tend to reveal a positive willingness to pay for risk reduction. But there is no particular reason why their valuations of risk should be the same;

2. incomes vary and hence willingness to pay is likely to vary in such a way that those with higher incomes have higher WTPs. This is not a necessary result since attitudes to risk may vary in such

a way as to offset an income effect. Nonetheless, it raises an important equity issue about fairness between people, an issue that is not in fact confined to risk valuations but to the use of WTP measures in general.

A VOSL can also be measured by a ‘willingness to accept’ compensation for increased risk. It is well known that many people do make this trade-off between risk and money, for example by accepting premia on wages to tolerate risk. It is tempting to think that the WTA approach will produce very much higher values for a VOSL than the WTP approach, simply because WTA is not constrained by income. WTP and WTA can, indeed, be different and WTA for environmental losses may exceed WTP for environmental gains by factors of 2-5 (Gregory, 1986). Various explanations exist for this disparity, including the fact that individuals may feel they are losing an ‘entitlement’ if the issue is one of loss of an entitlement (WTA) rather than an increment to an existing entitlement (WTP). Another explanation, which is wholly consistent with economic theory, suggests that WTA > WTP arises mainly in contexts where there is no ready substitute for the environmental good in question (Hanemann, 1991).

Table 3.1 presents the main valuation techniques used for estimating VOSL and the types of risks they measure based on the literature to date. Use of VOSL estimates of the kind noted in *Table C1* has come under criticism for several reasons:

- there is unease about the fact that health benefits based on VOSL are so dominant in cost-benefit studies;

- the VOSL estimates come largely from accident contexts where the mean age of the person killed is very much lower than in pollution contexts. There is therefore a feeling that older people, perhaps with an already impaired health state will not have the same valuation of risk as someone who is very much younger; and

- it is, as noted above, very easy to confuse what a VOSL is actually measuring. Wrongly translated as a ‘value of life’, the concept is easy prey for critics who do not invest in attempts to understand the analytical foundations of VOSL. Since this confusion is widespread, analysts often prefer not to use the VOSL concept at all.

Table 3.1 Valuation techniques for measuring the risk to human health

Method	Scope	Comments
Hedonic wage (WTA)	Comparison of wage premium between occupations with different levels of risks to human health. Values from literature range between £2.4 and 4.6 million	Other variables such as education, experience, the existence of unions affect the results; perception of risk is more important than the actual level of risk; evidence suggests that workers overstate the risk of their jobs
Avertive behaviour (WTP)	Measures the actual expenditure on smoke alarms, safety harnesses, tamper-proof drug storage containers and other products that reduce the risk to human health. Values from literature range between £0.5-2.8 million	In practice, finding examples of avertive expenditure that are purely to avoid risk to health has provide difficult
Contingent valuation (WTP and WTA)	Respondents are asked how much they are WTP or WTA to avoid or suffer a marginal change in the risk of dying. Values (WTP/A) from the literature range between £0.4-5.3 million	The responses are found to depend on the level and type of risk and whether or not the respondent is in a risk group in addition to conventional factors such as income and education. Evidence shows that involuntary risk such as cancer is valued more highly than voluntary risk such as traffic accidents. Yet the empirical VOSL literature is almost entirely based on accident risks. For pollution issues, then, transferring VOSLs from accident risk contexts to pollution contexts is likely to understate the 'true' degree of risk aversion.
Human capital approach	What the person would have produced in the remainder of their lifetime net of their consumption – had they lived	The approach is inconsistent with WTP rule and is not used as extensively as before the WTP was developed

Of these reasons, only the second has any intellectual basis, although the first does reflect a 'statistical sensitivity' issue in the sense that, if the VOSL estimates are wrong, then entire decisions may be changed.

For these good and bad reasons, then, there have been attempts to estimate not the value of the risk of fatality but the value of the life period gained by reducing the risk. This has come to be known as the 'value of a life year' or VOLY.

Value of a Life Year (VOLY)

The underlying rationale for valuing ‘life years’ is that many contexts in which health risks occur relate to pollution. Clearly, pollution is more likely to affect people who are most vulnerable. In a poor country this may be the very young and the very old. In a rich country, where infant mortality risks are very low, it is more likely to affect the elderly and especially those who are already at risk from their prevailing health state. Suppose, for argument's sake, that, statistically, the reduced life expectancy of someone exposed to air pollution is six months. Then, the argument goes, what matters is the value the individual places on those six months of extended life. If the period is a few weeks or even days, then the relevant value is that 'life period' rather than the actual risk.

This contrasts with the VOSL where a person, however old they are, is faced with a risk and they express their WTP to reduce that risk. In principle, the two values - VOSL and VOLY - should bear some relationship since the person at risk must have some idea of remaining life expectancy. Indeed, it would be extremely surprising if they did not. In expressing a WTP to reduce risk, then, they should be accounting for the remaining life period available to them.

One obvious way of approaching the problem is to see if WTP to reduce risks is functionally related to age, an issue we return to below. The surprising thing about the VOSL literature is that very little of it controls for age, so that only a few studies exist to offer a guide on how risk valuations vary with age.

Alternative approaches attempt to estimate the VOLY and, so far, two procedures have been used. The first simply takes estimates of the VOSL and converts them to values of life year; i.e. no additional information is sought. The second attempts to construct VOLY from first principles by engaging in valuation studies that directly attempt to elicit the WTP for extended periods of life.

VOLY derived from VOSL

One approach to estimating the VOLY is to regard it as the annuity which when discounted over the remaining life span of the individual at risk would equal the estimate of VOSL.

$$\text{VOLY} = \text{VOSL}/\text{A}$$

where $A = [1-(1+s)^{-T}]/s$ and T is years of expected life remaining and s is the utility discount rate[20].

While the VOLY approach may appear sound, it offers no evidence that VOSL declines with age in the manner shown. If this were to be the case, we would expect to find evidence that the WTP to reduce risks varies inversely with age. As Rowlatt et al. (1998) note, there *is* some evidence for a declining WTP as people become older, but that evidence is not at all consistent with the age profile of VOSL as dictated by the VOLY approach. Maddison (1998) suggests that there are sound reasons for supposing that VOSL is proportional to the number of discounted life years remaining to an individual and that it is inversely proportional to the survival probability in the current time period. In other words, Maddison suggests that there are rationales for a declining VOSL with age, but that this will be attenuated in old people by the reduced survival probability. For the UK, he suggests that the VOSL for a 74-year-old with six months life expectancy would be 17% of the healthy 36-year-old.

However, to quote a study for the US Environment Protection Agency:

'..it is possible that the reduced life expectancy and reduced enjoyment of life associated with many chronic illnesses may result in lower WTP to reduce risks of death. On the other hand, facing serious illness and reduced life expectancy may result in higher value [being] placed on protecting the remaining time.' (Chestnut and Patterson, 1994) .

Overall, Maddison's approach holds out some promise for finding age-related VOSLs via indirect routes. These should then be tested against VOSLs derived from direct approaches in which age is specifically accounted for.

[20] The utility discount rate is the rate at which future wellbeing is discounted, not the rate at which income or consumption is discounted.

VOLY derived from WTP experiments

An alternative procedure based on the VOLY concept is to see the WTP to extend a lifetime conditional on having reached a certain age. There is only one such study to date (Johannesson and Johansson, 1996) that reports a contingent valuation study in Sweden that measured people's preferences for one extra life year at the age of 85. The results suggest average WTP across the age groups of slightly less than 10,000 SEK using standard estimation procedures and 4,000 SEK using a more conservative approach. In dollar terms this is \$600-1500. Recall that this is for one year of expected life increase. WTP actually *increases* with age, although not dramatically - on the standard basis, 8000 SEK for the 18-34 age group, 10,000 for the 35-51 age group and 11700 for the 51-69 age group. Using the formula:

$$VOSL(a) = VOLY.\Sigma\ 1/(1+s)^{T-a}$$

Where T is life expectancy and a is the age at which VOSL is measured. Johannesson and Johansson suggest these values are consistent with 'normal' VOSL of \$30,000 to \$110,000, substantially less than the VOSL derived previously. Since T-a is obviously less the older the age group, then the relevant VOSL will decline with age. They also derive discount rates of 0.3% to 3.4% and these are invariant with age.

Finally, they argue that these lower valuations are consistent with findings in Sweden and the USA on social attitudes to allocating resources to life saving. Thus, Cropper et al (1994) found that survey respondents strongly favoured life saving programmes which save the lives of young people rather than old people. Earlier work by Johannesson and Johansson (1995a, 1995b) found that Swedish attitudes were similar, and that expectations about the future quality of life at old age play a significant role (regardless of what the actual quality of life is). The implications of the low WTP values for health care are hinted at in Johannesson and Johansson (1996): they observe that the VOSL values are 'negligible' compared to the costs of health treatment for the aged.

Is the WTP approach used in the Johannesson and Johannsson study consistent with the VOSL approach ? It is arguable that the 'goods' being valued are quite different: VOSL studies value risk and the VOSL is simply an aggregation of those individual valuations of risk. The WTP for a life year is not explicitly a value of risk, but a value of extending a life year once the respondent is assumed to reach a particular age. The Johannesson and Johansson paper could be argued to be more relevant for pollution control policy if the benefits of that policy are thought to accrue mainly to the elderly.

Conclusions

Few topics have proved so controversial as the 'value of statistical life'. In large part the controversy derives from unfortunate terminology, since what appears to be at stake is the 'value of life' itself. This confusion has not been helped by even the most distinguished commentators and analysts using this phrase. But what is being estimated is the value of risk reduction. VOSL is essentially, a convenient way of aggregating these estimates.

In a finite world there really should be no dispute that resources have to be allocated rationally across different life risks. The real focus of the debate should be on the size of the VOSL. As we saw, this is the subject of a debate, which centres on two approaches to valuing risks. The first asks for the WTP to avoid risks, and the second asks for the WTP to extend an expected lifetime by some finite period, say, one year. The literature on 'value of life year' turns out to be a hybrid of these approaches, deriving VOLY from a given VOSL. As discussed, there appears to be limited theoretical justification for this hybrid approach. It is also not consistent with what we know about VOSL as they vary with age. Nonetheless, what we know about the age-WTP relationship is not much. In turn, the literature that attempts directly to estimate VOLY is minute. Such as it is, it suggests VOSL are very much less than those derived from standardised value of risk calculations.

Other issues concern the role that others' valuation of risks should play and the role that discounting might play in valuing future risks. In general it would appear that there is a case for adding a modest premium to own VOSL for others' altruistic concerns, and there is no strong case for discounting future risks.

Finally, note that majority of the studies estimating the economic cost of air pollution damage to human health that are reported in this study use the VOSL approach.

4. Monetary valuation of morbidity effects

Unit damage values for impacts to human health used in this study are taken from CSERGE et al (1999). For transparency, the disaggregated estimates are presented here. *Table 4.1* presents descriptions of the epidemiological end-points caused by air pollution and considered in the analysis. The unit willingness to pay to avoid each type of incident of ill-health is also presented. These values are specific to the Netherlands.

Table 4.1 WTP to avoid episodes of ill-health (Euro,2000)

Epidemiological End Point	Description	**WTP**
Mild symptom day	One day with mildly red, watering, itchy eyes. A runny nose with sneezing spells. Patient is not restricted in their normal activities.	61
Minor restricted activity day	One day with persistent phlegmy cough, some tightness in the chest, and some breathing difficulties. Patient cannot engage in strenuous activity but can work and do ordinary daily activities.	42
Work loss day	One day of persistent nausea and headache, with occasional vomiting. Some stomach pain and cramp. Diarrhoea at least twice during the day. Patient is unable to go to work or leave the home, but domestic chores are possible.	53
Respiratory bed day	Three days with flu-like symptoms including persistent phlegmy cough with occasional coughing fits, fever, headache and tiredness. Symptoms are serious enough that patient must stay home in bed for the three days.	107
Emergency room visit for COPD and asthma	A visit to a hospital casualty department for oxygen and medicines to assist breathing problems caused by respiratory distress. Symptoms include a persistent phlegmy cough with occasional coughing fits, gasping breathing even when at rest, fever, headache and tiredness. Patient spends 4 hours in casualty followed by 5 days at home in bed.	194
Hospital admission for COPD, pneumonia, respiratory disease and asthma	Admission to a hospital for treatment of respiratory distress. Symptoms include persistent phlegmy cough, with occasional coughing fits, gasping breath, fever, headache and tiredness. patient stays in the hospital receiving treatment for three days, followed by 5 days home in bed.	428

COPD = Chronic Obstructive Pulmonary Disease
* no estimate given for the Netherlands, so this estimate is taken from the pooled sample of all 5 European countries.
Source: CSERGE et al (1999).

In order to obtain an estimate of the total costs of episodes of ill-health, costs to employers and to the health system must also be considered. These are presented in *Table 4.2* below. Again, all values are specific to the Netherlands.

Table 4.2: Productivity and health service costs for ill-health (Euro, 2000)

Per diem productivity cost (Euro/day)	64
Emergency room visit (Euro/visit)	45
Hospitalisation (Euro/inpatient day)	367

Source: CSERGE et al (1999).

Table 4.3 uses the values presented in *Tables 4.1* and *4.2* above to derive the total cost of an episode of ill-health of each given type.

Table 4.3 Total costs of episodes of ill-health (Euro, 2000)

Epidemiological End Point	WTP	Productivity costs	ERV costs	Hospitalisation costs	Total costs
Symptom day	61	-	-	-	61
MRAD	42	-	-	-	42
Work loss day	53	64	-	-	117
Respiratory bed day	107	192	-	-	299
Emergency room visit	194	320	45	-	559
Hospital admission	428	320	-	367	1,115

Source: CSERGE et al (1999).

In the sensitivity analysis, the effects of using different values for these epidemiological endpoints is tested. Monetary valuation estimates are taken from ExternE (1998). These are summarised in *Table 4.4* below.

Table 4.4 Monetary valuation estimates used in the sensitivity analysis

Impact	Unit value (Euro, 2000)	Reference (estimation method)
Mortality 0 – 65 years	4.98	CSERGE et al, 1999
Mortality +65years	3.49	adjusted from CSERGE et al, 1999 (using results of Pearce, 1997)
Respiratory hospital admission (RHA)	8,304 – 8,853	Maddison, 1997 (adjusted COI) - Markandya, 1997 (CV)
Emergency room visit (ERV)	42 – 251	Maddison, 1997 (adjusted COI) – Markandya, 1997 (CV)
Restricted activity day (RAD)	20 – 85	Maddison, 1997 (adjusted COI) – Markandya, 1997 (CV)
Respiratory symptoms (SD)	5 - 9	Maddison, 1997 (CV) – Markandya, 1997 (CV)

5. Environmental data, assumptions and models

For the benefit analysis we used the most recent data available. We present data for the current status of each environmental issue (1995) and the expected future progress (2010, 2020 and 2030). Data are expressed as tonnes of emissions of each pollutant, concentrations and/or exposure levels. The data are drawn from the (draft) Fifth National Environmental Outlook for the Netherlands (NEO5). This report is prepared for the Fourth Dutch Environmental Policy Plan (to be published in the beginning of 2001).

The aim of NEO5 is to assess the state of the environment in the view of sustainable development in the Netherlands and in its international context. The NEO5 includes an appraisal of the nature and causes of the persistent environmental problems in the Netherlands, and it inventorises new emerging issues. The appraisal of the persistency of environmental problems is based on historical environmental data in relation to current policies (for the period 1970-2000) and also on future developments.

The following persistent environmental problems can be listed:

- climate change;
- acidification including tropospheric ozone;
- particulate matter;
- eutrophication, and
- noise disturbance.

Scenario and current legislation

Future progress of the environmental issues has been based on future societal trends. As these trends are in principal uncertain we have to apply scenarios. For this study we have only used one scenario, as the focus of the study is on ranking of the environmental issues. This ranking is unlikely to be affected by the scenario choice. For our study we have chosen the so-called European Co-ordination scenario (CPB, 1997). Characteristics of the scenario are the following:

- the rise of 'Fortress America' and the tendency toward strategic trade and industrial policies significantly contribute to formation of trade blocks;

- despite increasingly strained relations with the USA, Western Europe develops very favourably. The European process of integration is an important stimulus toward strengthening incentive structures on the Western European product and labour markets. A far-reaching process of reform of the West European welfare state is set in motion. In this, attempts are made to combine the European tradition of social equity, with an increased sensitivity to economic incentives;

- the EU introduces an energy/carbon tax of $ 10 per barrel;

- Technological development and diffusion is moderate;

- There is high migration to the EU.

Some key figures of EC scenario are presented in *Table 5.1*.

Table 5.1 EC scenario key figures

	1995	2020
Population (million)	15.5	17.7
Number of households (million)	6.5	4.5
Consumption per capita (index)	100	170
GDP growth (index)	100	195
Oil price ($ per barrel)	17	25

The societal trends of the EC scenario have been combined with the environmental policies already in place in the Netherlands[21] and the EU. This is referred to as the 'current legislation scenario' (CLS).

The following key environmental policies are included in CLS:

- Climate change policy plan (1999); implementation of Kyoto protocol;
- European emission directives (e.g., EURO IV);
- Most recent emission standards for combustion;
- Integrated policy plan for reducing ammonia and manure , and
- Most recent noise standards for transport.

Models and data flows

The results of the Fifth National Environmental Outlook will be published in July 2000. All environmental data (indicators) used in this report will be fully documented in so-called factsheets. These factsheets will contain the following:

- data source (RIVM or other);
- data consistency check;
- description of the indicator;
- computation (model name, version, spreadsheet);
- data uncertainty;
- data input and assumptions, and
- references.

The justification of the data input for the benefit estimations can be found in these factsheets. The annex of this report shows the main data flows of the NEO5 (in Dutch), (see *Annex I.6*).

General data used in this study are presented in *Table 5.2*.

Table 5.2 General data used in this study

Data	1995	2010	2020	2030
Population: millions	15.424	16.821	17.673	18.406
Size of households	2.34	2.27	2.23	2.21
GDP: Euro billion	312.6273	482.5431	632.9282	830.1809

Environmental data

The following tables present the environmental data used in this study. The data are provided by RIVM and are taken from the 5th National Environmental Outlook, (draft report), with the exception of land contamination, which is not discussed in that report.

[21] Policies in place as approved by the Dutch parliament before January 1, 2000

Climate change
Table 5.3 presents the greenhouse gases emitted from the Netherlands.

Table 5.3 Gases contributing to climate change: current legislation scenario

Carbon dioxide, CO_2: million tonnes				
	1995	*2010*	*2020*	*2030*
waste management	1.5	2.0	2.0	2.0
Construction	0.7	0.8	0.8	0.8
Households	21.8	21.5	20.0	18.3
water	0.0	0.0	0.0	0.0
energy	45.9	47.5	50.0	57.1
Services	9.9	13.5	15.3	16.7
Industry	44.3	45.2	49.3	53.7
Agriculture	9.4	10.4	11.0	11.6
Other	2.5	3.0	3.0	3.0
Refineries	11.5	15.6	17.0	17.8
sewage treatment	0.1	0.1	0.1	0.1
Transport	32.4	38.7	44.5	52.2
Total	180.2	198.3	213.1	233.3
Methane, CH_4: thousand tonnes				
	1995	*2010*	*2020*	*2030*
waste management	479	217	102	51
Households	18	16	16	16
energy sector	178	73	38	38
Industry	8	5	5	6
Agriculture	479	385	368	343
Other	5	5	5	5
Transport	6	3	2	2
Total	1172	703	537	462
Nitrous dioxide, N_2O: thousand tonnes				
	1995	*2010*	*2020*	*2030*
energy sector	0.4	0.2	0.1	0.1
Industry	31.7	32.9	34.2	35.8
Agriculture	27.6	20.9	20.0	19.6
Other	5.0	5.0	5.0	5.0
Transport	7.1	2.6	2.2	2.6
Total	71.9	61.6	61.5	63.1
Total per gas (million tonnes CO_2-eq)				
CO_2	180.2	198.3	213.1	233.3
CH_4	24.6	14.8	11.3	9.7
N_2O	22.3	19.1	19.1	19.6
HFK	6.7	3	5.4	6.5
PFC	2.1	0.9	1.1	1.2
SF6	1.5	2.0	2.3	2.7
Total (IPCC)	*237.3*	*238.1*	*252.2*	*273.0*

Source: Fifth National Environmental Outlook (draft report).

Acidification

Table 5.4 presents the acidifying pollutants emitted from the Netherlands.

Table 5.4 Gases contributing to acidification: current legislation scenario

Ammonia, NH_3: 1000 tonnes				
	1995	*2010*	*2020*	*2030*
Households	6.7	7.4	7.8	8.2
Industry	4.1	4.1	4.3	4.7
Agriculture	176.9	143.0	133.4	128.0
Other	0.6	0.7	0.8	0.8
Total	188.4	155.2	146.3	141.7
Nitrogen oxides, NO_x: 1000 tonnes				
Households	22.3	14.4	12.0	9.8
Energy	58.1	23.8	25.4	25.4
Services	8.1	7.6	7.4	7.0
Industry	61.2	42.5	42.2	42.2
Agriculture	10.3	12.0	9.5	7.5
Other	3.8	2.4	2.6	2.6
Refineries	17.7	8.1	8.2	8.2
Transport	313.1	161.2	149.3	173.5
Total	494.6	272.1	256.6	276.2
Sulphur dioxide, SO_2: 1000 tonnes				
Waste	0.5	0.3	0.3	0.3
Energy	16.7	9.8	8.6	10.5
Industry	29.8	23.8	24.7	25.7
Other	5.7	2.7	2.8	2.8
Refineries	61.2	21.0	22.0	22.3
Transport	29.9	13.7	15.4	17.2
Total	143.8	71.2	73.8	78.6

Source: Fifth National Environmental Outlook (draft report).

Low level ozone

Table 5.5 presents VOC emissions that contribute to low level ozone in the Netherlands. NO_x emissions are given in *Table 5.4. Table 5.6* reports the ill-health incidences associated with exposure to low level ozone. The dose response functions used to derive the ill-health incidences are reported in the appendix to *Section 4.7*.

Table 5.5 VOC emissions: current emissions scenario: 1000 tonnes

Source	*1995*	*2010*	*2020*	*2030*
Transport	148.3	51.9	48.4	52.4
Industry	80.9	55.4	59.5	63.6
Households	39.6	31.1	33.7	37.7
Services	29.2	18.7	20.8	23.1
Energy	26.2	15.8	13.8	13.8
Construction	21.9	10.5	11.8	12.1
Refineries	11.7	9.0	9.0	9.0
Other	4.0	2.4	2.2	2.1
Total	*361.8*	*194.9*	*199.2*	*213.7*

Source: Fifth National Environmental Outlook (draft report).

Table 5.6 Mortality and morbidity effects due to ozone

Mortality: number of people				
	1995	*2010*	*2020*	*2030*
Total	1,540	2,078	2,500	3,246
Aged under 65	110	147	153	141
Aged 65+	1,448	1,962	2,354	3,226
Morbidity: number of hospital admissions				
Total	449	619	749	869
COPD	281	389	472	548
Asthma	0	0	0	0
Cardiovascular d.	168	230	278	320

Source: Fifth National Environmental Outlook (draft report).

Noise

Table 5.7 presents the percent population in various noise bands.

Table 5.7 % population in various noise bands: dB(A)

1995	**0 – 40**	**41-45**	**46-50**	**51-55**	**56-60**	**61-65**	**66-70**	**71-75**	**76-80**	**>80**
Total	5.24	6.61	16.93	29.17	27.03	11.68	2.76	0.47	0.10	0.01
Road transport total	6.84	9.76	22.40	28.45	21.96	8.70	1.71	0.17	0.02	0.00
- high ways	52.77	13.40	16.77	11.12	4.27	1.20	0.34	0.11	0.02	0.00
-regional roads	57.35	16.75	12.93	7.07	3.45	1.74	0.69	0.03	0.00	0.00
-city roads	28.68	16.94	19.23	16.87	12.76	4.98	0.55	0.00	0.00	0.00
Rail	57.89	14.22	13.06	8.06	3.85	1.80	0.77	0.26	0.08	0.01
Airport	81.30	3.40	8.99	4.72	1.38	0.17	0.03	0.00	0.00	0.00
2010										
Total	4.29	5.21	13.69	28.19	29.65	14.22	3.84	0.74	0.15	0.02
Road transport total	5.98	8.35	20.55	28.85	23.37	10.17	2.41	0.30	0.02	0.00
- high ways	52.35	12.55	16.63	12.27	4.44	1.23	0.39	0.11	0.02	0.00
-regional roads	52.31	16.96	14.52	8.67	4.28	2.09	1.04	0.14	0.00	0.00
-city roads	28.38	15.25	19.20	17.40	12.95	5.94	0.86	0.02	0.00	0.00
Rail	54.71	13.23	13.54	10.11	4.78	2.13	0.98	0.37	0.12	0.02
Airport	68.98	8.40	10.31	8.70	3.19	0.36	0.05	0.01	0.00	0.00
2020										
Total	4.30	5.03	12.96	26.66	30.44	15.15	4.36	0.91	0.16	0.04
Road transport total	6.14	7.87	19.21	28.15	24.33	10.90	2.89	0.47	0.03	0.00
- high ways	51.26	11.42	16.26	13.34	5.44	1.62	0.48	0.16	0.03	0.00
-regional roads	49.48	16.89	15.34	9.60	4.84	2.40	1.21	0.24	0.00	0.00
-city roads	30.61	14.51	18.38	17.00	12.55	5.86	1.05	0.03	0.00	0.00
Rail	57.10	12.24	12.66	9.61	4.84	2.11	0.94	0.36	0.11	0.03
Airport	67.70	8.82	9.82	9.24	3.81	0.49	0.11	0.01	0.00	0.00
2030										
Total	3.82	4.49	11.94	25.86	31.42	16.25	4.88	1.13	0.19	0.04
Road transport total	5.43	7.01	17.79	27.87	25.90	11.83	3.45	0.68	0.05	0.00
- high ways	49.66	10.31	16.17	14.50	6.58	1.95	0.59	0.19	0.04	0.00
-regional roads	47.52	16.48	15.88	10.45	5.31	2.65	1.36	0.35	0.01	0.00
-city roads	29.77	14.24	18.30	17.42	12.83	6.03	1.33	0.08	0.00	0.00
Rail	58.26	12.34	12.10	9.16	4.65	2.09	0.88	0.35	0.12	0.03
Airport	66.13	8.97	10.26	9.74	4.16	0.60	0.12	0.01	0.00	0.00

Land contamination

Table 5.8 gives the number of contaminated sites in the Netherlands in and their distributions amongst land use types (1996 figures) and the distribution of land uses when all contaminated sites are cleaned. The table also shows the size (ha) of contaminated sites both for 1996 (all contaminated) and for the time when all contaminated land is cleaned.

Table 5.8 Contaminated areas

	Contaminated Land		Cleaned Land	
	No. of sites	Land area (ha)	No. of sites	Land area (ha)
Total	351,000	631,800	351,000	631,800
Residential	91,228	164,210	150,000	269,993
Industrial	126,328	227,390	90,768	163,384
Agricultural	41,084	73,953	39,430	70,970
Recreational and nature	23,292	41,926	26,676	48,017
Derelict	29,600	53,280	4,696	8,453
Dumping	7,925	14,267	2,179	3,924
Other	31,540	56,774	37,248	67,046

Source: RIVM, 1997 Achtergronden bij: Milieubalans 1997, Samson HD, Tjeenk Willink Alhen aan de Rijn.

Particulate matter

Table 5.9 reports emissions of PM_{10}, whilst *Table E10* gives the associated ill-health incidences. The dose response figures used to determine ill-health incidences are given in the appendix to *Section 4.5*.

Table 5.9 PM_{10} emissions: current legislation scenario: 1000 tonnes

	1990	*2010*	*2020*	*2030*
Consumers	6.8	6.5	6.6	6.7
Energy sector	0.6	0.4	0.3	0.3
Industry	14.3	7.9	8.2	8.4
Other	1.3	0.7	0.7	0.2
Refineries	4.8	2.0	1.8	1.8
Transport	18.6	9.9	9.8	11.0
TOTAAL	46.4	27.5	27.4	28.4

Source: Fifth National Environmental Outlook (draft report).

Table 5.10 Mortality and morbidity effects due to PM_{10}: current legislation scenario

Mortality (number of people)				
	1995	*2010*	*2020*	*2030*
Total	931	992	1172	1529
Aged under 65	135	147	152	142
Aged 65+	796	845	1020	1387
Morbidity (number of hospital admissions)				
Total	3608	3597	4399	5320
COPD	958	953	1161	1401
Asthma	133	122	129	136
Cardiovascular disease	1214	1240	1576	1963

Source: Fifth National Environmental Outlook (draft report).

Eutrophication

Table 5.11 gives the nitrogen and phosphorus emissions from the Netherlands.

Table 5.11 Nitrogen and phosphorus emissions: current legislation scenario

N: 1000 tonnes				
	1996	*2010*	*2020*	*2030*
Total emissions	56	33	33	33
Domestic effluent	35.06	21.84	22.81	23.46
Agriculture	6.27	0.92	0	0
Industry	5.70	3.80	3.80	3.80
Atmospheric deposition	5.62	4.21	4.01	3.88
Sewage	1.20	0.98	0.93	0.88
Overflow	1.09	0.97	0.88	0.87
Households not connected	0.69	0.23	0.23	0.23
P: 1000 tonnes				
Total emissions	7.6	3.5	3.6	3.6
Industry	3.55	0.66	0.66	0.66
Domestic effluent	3.36	2.64	2.73	2.80
Overflow	0.12	0.09	0.11	0.11
Households not connected	0.11	0.07	0.07	0.07
Agriculture	0.44	0.03	0.01	0.00

Source: Fifth National Environmental Outlook (draft report).

6. Data flows of the 5th National Environmental Outlook

Environmental Pressures and the Calculation of Emissions
The main environmental pressures on health and nature are considered to be climate change, long-range air pollution (acidification and smog), eutrophication and urban stress (noise and urban air quality). Analysis of the environmental pressure exerted by the economic sectors relevant to these environmental themes is the chief responsibility of the RIVM, both for the annual Environmental Balances and for the Environmental Outlooks, making use of the relevant input data, scenarios, environmental policy and technological developments. The developments given in the socioeconomic scenarios are translated into emissions to the environment. *Figure 2.1* below shows the general features of this process.

The most basic formula for determining all emissions can be reduced to the following:

E = V (economy-, demography- and policy-related) * K (technology, policy)
where:
E = Emission
V = Volume (e.g. industrial production volume, transport volume in tonne kilometres, animal stock, etc.)
K = Emission factor per Unit Volume.

The formula above is used at all relevant sub classification levels of production sectors and consumption categories, adding up to total Dutch production and consumption figures on a national scale.

Volume indicator
The first perhaps most relevant step in the calculations is to derive the environmentally relevant volume-indicators *V*. Volumes of production or consumption are a function of economic and/or demographic parameters as given in the economic scenarios and human behaviour. This combination is sometimes performed using econometric models with underlying price elasticity studies etc., with the relevant volume-indicators as output results.

An example is the traffic model FACTS 3.0 (Forecasting Air pollution by Car Traffic Simulation). This flexible model, originally developed by the Netherlands Economic Institute, draws up forecasts of car ownership and use as well as their emissions and energy consumption, all under alternative scenarios of economic and demographic development, emission regulation, fuel efficiency, prices of fuels and cars, and other government measures relating to traffic and transport.

Many of the policy and other model parameters can be varied, for instance economic development, demographic evolution, developments in fixed and variable car costs, and levels of emission and energy consumption.

In other cases, specific data necessary for the calculation are not available. In that case the estimation and prediction of volumes is based on the historical relationship between monetary data and emissions. Where relevant, the influence of environmental policy is also taken into consideration, resulting in generally accepted restrictive measures. These measures involve limitation of the increase in livestock as a consequence of policy measures, the influence on traffic and transport, etc.

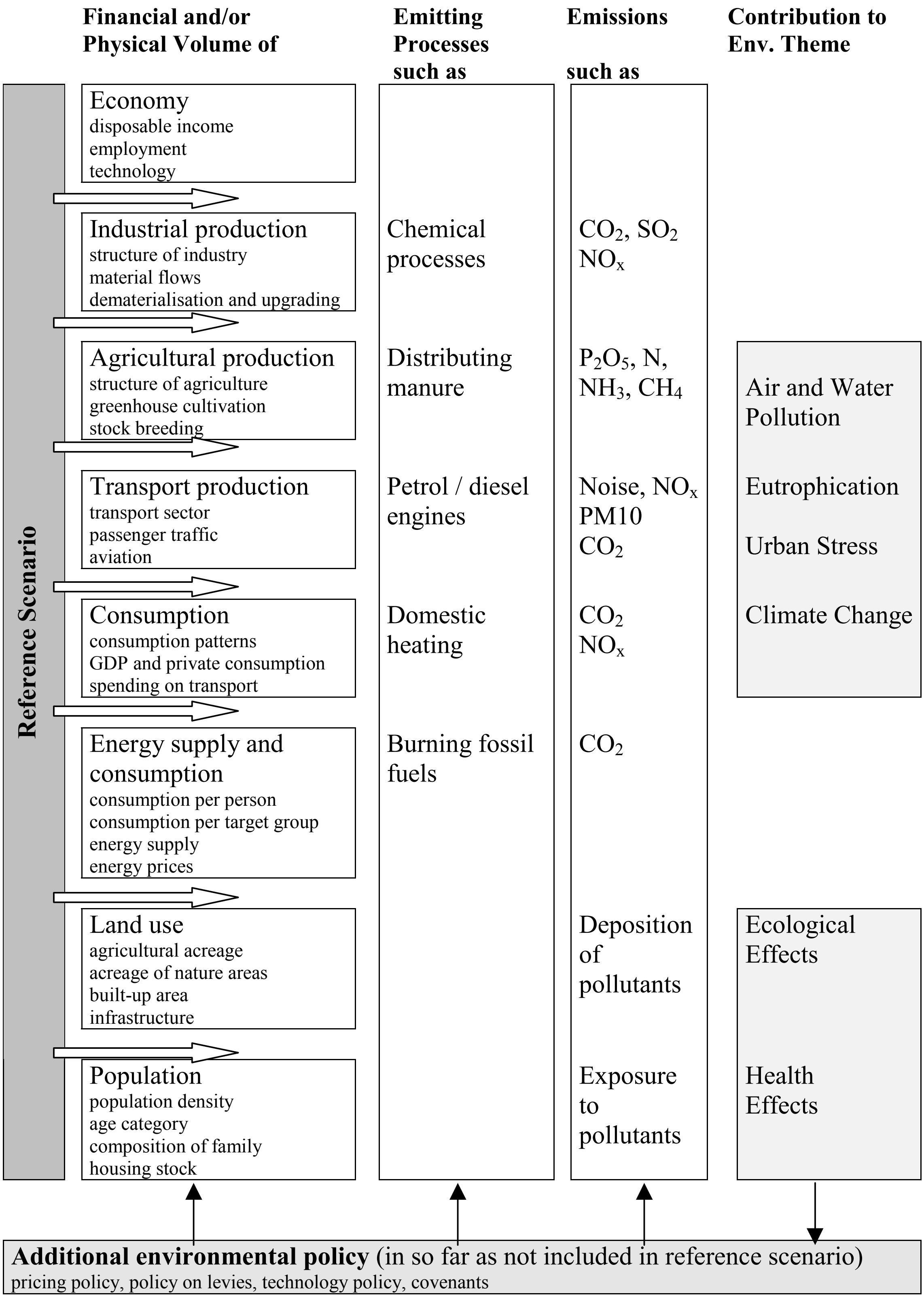

Figure 2.1 Chart for calculating emissions

Emission factors
Emission factors K (per Unit of Volume) are influenced by technological developments and the way these developments are affected by policy measures.

- Technological development

Technological development plays an important role in the link between physical production and resulting emissions and is seen by the Dutch government as an important instrument in improving environmental quality and avoiding economic harm. RIVM's inventory of technological developments and their impacts on emission factors is based on both studies of the literature and expertise in the field of application. Prognoses up to 2020 mainly involve conventional technology. Detailed information on technology and implementation grades is aggregated to annual averages. For prognoses from 2020 up to 2030 more aggregated assumptions are made and more radical changes and breakthrough technologies are taken into consideration.

- Environmental policy

The reference scenario in the 5th Environmental Outlook is based on all 'agreed policy measures'. 'Agreed policy measures' are defined as 'all government-endorsed policy instruments translated into real technological or other measures to be implemented in the various sectors of industry, agriculture, etc.'. A draft list of these instruments/measures is communicated to and discussed with policymakers to obtain consensus and avoid misunderstanding.
If certain agreed policy results receive very low public and/or interdepartmental support and implementation proves to be doubtful, margins in their environmental effects are given by presenting both calculation results.

- Specific

If no more widely accepted specific scenarios are available, e.g. on lifestyle, consumer behaviour, technical developments, etc., expert judgement teams are invited to participate in a broad debate in order to obtain the largest possible consensus for specific scenarios.

Uncertainties
In calculating trends in volumes and emission factors uncertainties cannot be avoided. Therefore, scenario data contain margins or a bandwidth. Future implementation grades e.g. have to be estimated, using knowledge of present state-of-the-art technology and the tendency to technological innovation in the sectors in the future.
It turns out that uncertainties in policy implementation can have as much impact on the outcomes as economic uncertainties, sometimes even more.
In addition to the reference scenario, additional policy options are defined in close collaboration with policymakers and their effects are estimated by calculation. The results obtained form the input for the National Environmental Policy Plan.

Annex II Integrating cost-benefit analysis into the policy process

by

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with

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1. Purpose of the paper

This paper is concerned with the issue of how to introduce more **rational procedures** into public decision-making. It is a non-technical paper, written in the belief that there is still scope to improve the ways in which politicians, civil servants, experts, advisers and the community at large currently make decisions. Exactly what is meant by 'improving' decision-making will become clear as we go along. The focus is on economics, improving decision-making by using economics and why economists have some special insights to offer. Economists are not alone in having thought about the principles of better decision-making. However, others will be better placed to give advice on how other disciplines can improve decision-making.

The paper is structured as follows:

(a) we begin with the logical structure of rational decision-making (RDM), focusing on an **alternatives-criteria matrix** familiar in policy analysis. The alternatives-criteria matrix is the basis upon which all interested parties can have discourse about decision-making. This discourse is the essential basis for introducing RDM into decision-making procedures. In particular, we focus on **cost-benefit analysis**. We argue that cost-benefit analysis (CBA) is a comprehensive RDM which does, however, focus on the notion of **economic efficiency**. In so far as other objectives are relevant to decision-making, CBA will be a necessary but partial input to comprehensive decision-making.

(b) getting a discourse under way also involves addressing **the baseline issue**, i.e. what would happen if a given RDM procedure is not used and some other procedure is used instead.

(c) countries differ in the extent to which they employ rational decision making procedures and it is an interesting question as to why this is the case. We address this issue also since it casts some light on how procedures for enhancing the use of RDM can be implemented.

(d) we also look at the issue of what to do if a given RDM cannot be implemented fully because of technical or data difficulties. In particular, we ask what should be done when monetised costs and benefits can be estimated for some, but not all, effects of a policy or a project.

(e) While there are many technical issues relating to the validity or otherwise of CBA, a particular one has been highlighted by the steering committee for this paper: how to set the **geographical boundaries** of costs and benefits. We address this issue directly.

(f) Finally, we look at some **strategic matters** relating to the process for introducing RDM procedures. Of necessity, strategy will vary with the political and institutional contexts, so generalisations are difficult. Nonetheless, we suggest there are some basic rules that might be followed by those who wish to advance RDM procedures in government.

We conclude that CBA is a powerful RDM tool. It is not without problems but the issue is whether those problems render its results invalid, and whether the results could be improved upon by adopting other RDM techniques. We argue that other techniques have as many, if not more, drawbacks as CBA.

2. The issue: how to introduce rationality in public decision-making

The first issue to be addressed is the **context for rational discourse** about how to make decisions. Unless the ground rules for having this discourse are agreed, most argument and advocacy will be fruitless. Experience with decision-making suggests that most people would agree certain ground rules, even if they would want to stress some criteria for RDM over others.

All public decision-making involves **choice**. The necessity of choice arises because public funds are limited. Resources allocated to one purpose cannot be used to fund a different purpose. Money is a measure of resource use. Money spent on, say, education, cannot be spent on health care or the environment. Hence choices, or **trade-offs**, have to be made. To the economist, the inevitability of making choices is what defines the subject matter of economics. The value of the option that is sacrificed when a particular choice is made is known as the **opportunity cost**. If A is chosen over B, the opportunity cost is the lost benefit from B. While it may seem redundant to remind decision-makers that all their decisions involve an opportunity cost, professionally-trained people often do not behave as if there is a sacrifice when they make choices. The reality is that they probably do appreciate there are opportunity costs. What they are more likely to be saying is that *their* use of the resources is more valuable to society than the alternative use of the resources. The librarian will argue that spending money on libraries is vitally important. The medic will invoke the value of saving lives to justify more spending on medicine. The media will express outrage that a particular drug is not available on demand simply because it is very expensive. The schoolteacher will speak of the special needs of children and of education for the future. Each may speaks of ‘rights’ to information, health, education and so on. But in a finite world, there are no absolutes. Some things are indeed very important, but nothing is so important that it justifies allocating all of society’s resources to it, nor does any society behave that way. Because the subject matter of **economics** is defined as making choices in the context of finite resources, economists have a lot to say about **how to choose** between alternative uses of resources. That does not mean that economists alone should give guidance on how to choose. All disciplines have something to offer in terms of this guidance, but it seems fair to say that economic advice is very important precisely because economists have a discipline which is defined by this issue of how to make choices.

To the economist, **rational choice** means making the **‘best’ use of available resources**. The focus is mainly on a nation’s resources, because those are the resources over which national decision-makers have some control. But the principles of rational choice are just as applicable to resources beyond national frontiers. By engaging in international agreements, for example, nations effectively make some claim to determining choices that relate to the resources of other nations. But what is the ‘best’ use of resources?

‘Best’ is a **value word**. That means that what is best depends on what the **objective** is that we choose to try and achieve. In turn, there can be many different, legitimate objectives of social and economic policy: to increase employment, protect the environment, give special attention to the vulnerable at home and abroad, stimulate technological change, protect future generations, and so on.

The first reason why individuals and agencies may dispute the use of a particular RDM is that they may have different objective functions - very simply, they are concerned to achieve different objectives.

We suggest shortly that a significant part of the discourse about RDM has to be about agreeing objectives. Even if one group advocates a given objective, it must recognise that others have objectives that are also likely to be legitimate.

Governments always have **multiple objectives** and there will usually be an **inconsistency of objectives** simply because achieving one objective will often mean having to relax progress in achieving another objective. Protecting the environment may not be consistent with, say, liberalising the electricity market because liberalisation usually means lowering energy prices as a result of increased competition. Yet lower prices encourage more energy consumption and hence more atmospheric emissions and hence more environmental damage. Even if strenuous efforts are made to protect the environment, these incompatible objectives make those efforts much more difficult to achieve. Many decision-makers are often tempted to avoid this 'fact of life'. They eagerly seek **win-win solutions**, actions which purportedly achieve all objectives simultaneously. While there is some scope for win-win solutions, the idea that there is vast scope for such policy measures is an illusion. **There are nearly always losers**, and policy therefore has to be designed either to tolerate the fact of losers, or to compensate losers as best as can be done. Indeed, the notion of opportunity cost is there to remind us that there will always be at least one group of losers, namely those who would have benefited from a policy measure that cannot now be implemented because resources have been used for some other purpose.

The second reason for disagreements over RDM is that there is often a reluctance to accept the existence of losers. Effectively, some decision-makers look for the illusory goal of win-win solutions and deny that difficult choices have to be made.

If it was possible to attach a number to each policy objective such that an increase in employment of 10,000 people was worth, say, five points, and a 10% reduction in NOx emissions was worth two points, and so on, we could try to achieve the 'best' result by getting the most points possible. For example, if a stimulus to industrial growth would result in increasing employment by 10,000 but would increase NOx emissions by 10%, then the net effect of the policy would be +5-2 points = +3 points. It would be worth the sacrifice of more air pollution to secure the employment gains. While the example is simplistic, this is essentially what **cost-benefit analysis** does. Cost-benefit (CBA) attaches a score to the change in the indicator associated with each objective and then adds those scores up. If the net change is positive, the policy (or project or programme) is deemed socially worthwhile. If it is negative it is not worthwhile. This scoring procedure would be an example of rational decision-making provided we have some justification for using a particular scoring procedure. If the five points given to employment are purely arbitrary, then the procedure cannot be called rational. CBA adopts a particular rationale for scoring gains and losses.

Once it is accepted that there are always trade-offs, some form of weighting has to be adopted. It can be implicit, or explicit. In the interests of governmental transparency it should be explicit. CBA offers a procedure for adopting explicit weights based on people's preferences.

CBA derives the scores from people's preferences. Because it is so important, it is worth spending just a little time on how this is done.

In economics, benefits and costs are defined in terms of individuals' preferences. An individual receives a **benefit** whenever he/she receives something in return for which he/she is willing to give up something else that he/she values. To measure how large that benefit is, we measure how much she is willing to give up to get it. Conversely, an individual incurs a **cost** whenever she gives up something that she would willingly give up only if she was given something else that she valued as **compensation**. To measure how large that cost is, we measure how much would compensate her for incurring it.

These formulations define benefits and costs in terms of one another. The measure of any benefit is that cost which, in terms of the preferences of the individual who benefits, would exactly offset it. And, conversely, the measure of any cost is that benefit which, in the relevant individual's preferences, would exactly offset it. This is not circular. It reflects a crucial feature of economic evaluation: **there is no absolute measure of value**, there are **only equivalences of value between one thing and another**. By not claiming that any particular dimension of human life – health, material wealth, happiness, achievement or whatever – has absolute value, economic valuation avoids taking any substantive position about what is good for people. It simply uses whatever relative valuations are revealed in people's preferences. **Individuals' preferences are said to be sovereign**.

It is important to understand the notion of individual sovereignty because a decision-maker may wish to reject this basic value judgement. If he or she does so, then, while the basic idea of comparing gains and losses may remain, the structure of economic cost-benefit analysis will not be relevant. We return to this point later.

This approach allows all costs and benefits to be measured in a single dimension if, *as a matter of convention*, we choose one particular type of benefit to use as a standard. We can then express all other benefits and costs in terms of that standard, using individuals' own preferences to determine equivalences of value.

If we are to use the same standard of measurement for all individuals, the standard has to be a **good that everyone prefers to have more of rather than less**, and that individuals treat as a potential substitute for the array of benefits and costs that we want to measure. ('Substitute' is here used in a subjective sense: with respect to the preferences of a given individual, two goods are substitutes for one another to the extent that that individual is willing to accept a gain of one as compensation for a loss of the other.) And it has to be **finely divisible**. In economics, the usual convention is to use **money as the standard of measurement**. Money, obviously, is finely divisible. It represents general purchasing power – that is, the power to buy from the vast range of goods that are sold on markets. Because money can be put to so many different uses, it is a safe generalisation that most people prefer more money rather than less, irrespective of their specific preferences among goods. For the same reason, money is a particularly effective substitute good.

Using money as the standard of measurement is analytically very powerful. It is also a source of problems because money has all kinds of social connotations, e.g. with 'greed', 'avarice' and even 'evil'. Allowing this 'picture thinking' to dictate the choice of an RDM is clearly undesirable, so that the arguments for using money as a measuring rod must be presented carefully.

If money is used as the standard, the measure of benefit is **willingness to pay** (WTP). That is, a benefit to any given person is measured by the maximum amount of money that that person would be willing to pay in return for receiving the benefit. Similarly, the measure of cost is **willingness to accept compensation** (WTA). That is, a cost to any person is measured by the minimum amount of money that that person would be willing to accept as compensation for incurring the cost.

These measures of benefit and cost underlie the concept of **economic efficiency**. A reallocation of resources increases economic efficiency if the sum of the benefits to those who gain by that reallocation exceeds the sum of the costs to those who lose. In other words, there is an increase in economic efficiency if the sum of WTP for the gainers exceeds the sum of WTA for the losers. Another way of saying this is that there is an increase in economic efficiency, if (in principle) the gainers could compensate the losers without becoming losers themselves. This test is the **efficiency criterion** (or **compensation test**). CBA uses this criterion to appraise specific proposals. Note that economic efficiency is entirely compatible with there being actual losers, i.e. people who lose from the policy (project etc.) and who are not then actually compensated. All that is required for efficiency is that the beneficiaries *could* compensate the losers without themselves becoming losers.

It is a common error to think that the concept of economic efficiency refers only to some restricted 'economic' realm, and that a policy which is economically efficient is 'good for the economy' in a sense which is parallel to such judgements as that a policy is 'good for health', 'good for safety' or 'good for the environment'. Another version of the same error is to think that economic efficiency is about 'wealth creation', distinguished from activities which use the wealth supposedly created in the economy to buy valuable but non-economic goods such as health or environmental quality.

A related virtue of CBA, and of economic valuation more generally, is that it highlights the crucial concept of **opportunity cost**. In public debate, it is easy to express the judgement that some potential outcome is of great value without having to consider what might have to be given up in order to bring it about. Thus, for example, it is possible for there to be an apparent consensus in favour of each component of a package of public spending proposals while there is an equally evident consensus opposed to the tax increases which would be needed to pay for those proposals. Obviously, any sensible technique of decision analysis will identify the costs of options *to decision-makers*. But CBA does more than this. As data on the values of costs and benefits, the only information it accepts is information about preferences – that is, about *citizens' own willingness to substitute one thing for another*. Thus, in order for any potential outcome to be registered as having economic value, it is a precondition that the beneficiaries are willing to give up other valuable things in order to obtain it.

We now have an answer to our question about what is the 'best' use of resources. **The best use of resources is that use which maximises the net benefits to individuals in the defined community (nation, region etc.)**. Note that this is one definition. Because 'best' is a value word it is open to anyone to declare that a use of resources that meets some other object is the best use of resources. Ultimately, it is hard to deliberate between different definitions of the best use of resources because there is no absolute moral standard that tells us how to choose between alternative value systems. Probably the most important feature of the CBA approach is that it is '**democratic**'. Unless there are very good reasons to the contrary, resources should be allocated so as to meet what people want.

3. The criteria / alternatives matrix

The essence of the trade-off issue can be demonstrated using a **criteria/alternatives matrix**. This is illustrated in *Table 3.1*. The alternative policy options are shown at the top right of the matrix, say, changes in a speed limit on major roads. The criteria by which the desirability of changes in the speed limits could be judged are shown to the left of the matrix. The criteria might be number of serious accidents, travel time saved and the cost of operating vehicles. The cells of the matrix then show estimates of the change in speed limits on each of the criteria used. Illustrative numbers are provided. The second matrix shows the same information but this time 'normalised' on an existing reference base, e.g. the current speed limit. Since injuries and cost increase with higher speed, but time spent travelling falls, there is a trade-off between the criteria.

Tables 3.1 and *3.2* make it clear that the choice of the 'right' speed limit depends on factors over and above the 'basic' information provided about the effects of speed limits. What is required is some mechanism for trading off the time, cost and injuries impacts, i.e. we need to know at what rate the benefits of saving time can be traded for increased cost and increased injuries. Economists adopt preferences, as revealed through willingness to pay, as the mean of making the trade off. To illustrate, *Table 3.3* shows what happens if we adopted the following WTP Tables: each injury is valued at 1 million Dfl and a year of saving time is also valued at 1 million Dfl. Then the computation is simple, as *Table 3.3* shows. The highest net benefit would be for a change in the speed limit to 70 km/hr. Again, note that this result is obtained by taking people's preferences into account and using the rates of trade-off that people have indicated. Provided there is some sound rationale, the matrix in *Table 3.3* could also be generated by *any* procedure which produces rates of trade-off.

Table 3.1 Criteria/Alternatives matrix: original data

Criteria	70 km/h	80 km/h	90 km/h	100 km/h
Serious injuries per million vehicle kms	5.0	5.4	5.9	6.5
Time spent travelling: years per million vehicle kms	3.3	3.1	2.9	2.7
Vehicle operating costs per million vehicle kms (mill.Dfl)	12.6	12.8	13.2	13.8

Table 3.2 Criteria/Alternatives matrix: normalised on 80km/h

Criteria	70 km/h	80 km/h	90 km/h	100 km/h
Serious injuries per million vehicle kms	-0.4	0	+0.5	+1.1
Time spent travelling: years per million vehicle kms	+0.2	0	-0.2	-0.4
Vehicle operating costs per million vehicle kms (mill.Dfl)	-0.2	0	+0.4	+1.0

Table 3.3 Criteria/Alternatives matrix: monetised outcome (million Dfl)

Criteria	70 km/h	80 km/h	90 km/h	100 km/h
Money value of serious injuries	+0.4	0	-0.5	-1.1
Money value of time spent	-0.2	0	+0.2	+0.4
Money value of vehicle operating costs	+0.2	0	-0.4	-1.0
Aggregated net benefit	+0.4	0	-0.7	-1.7

Note: Benefits over the reference value of 80 km/h are shown as positive figures, while costs are shown as negative figures.

4. Summary so far

The discussion in Section 1 to explain the underlying 'philosophy' of CBA is essential if we are to understand why there is sometimes resistance to the use of CBA, and if we are to appreciate what measures we can take to encourage its wider use.

We can summarise the essential features of the discussion so far:

1 Decision-making is about making choices
2 We need a structured, rational approach that helps us decide how to choose
3 Rationality is about the best use of resources
4 'Best' is a value word
5 What is best will vary with the objective
6 Governments have multiple and often inconsistent objectives
7 Objectives have to be traded off
8 Some scoring methodology helps secure rational trade offs
9 CBA uses individuals' preferences as the scoring technique
10 Those preferences are revealed in willingness to pay and accept
11 Money is a convenient measuring rod
12 Individuals' preferences are assumed to be sovereign

5. What if all costs and benefits cannot be monetised?

CBA's answer to the problem posed by the crieria/alternatives matrix is to adopt money values as the weights to secure the outcome embodied in *Tables 3.1* to *3.3*. In turn, those money values have a very special meaning: they are measures of individuals' willingness to pay for the features that have to be traded off. Finally, willingness to pay reflects individuals' preferences.

One obstacle to the use of CBA is that it is not often possible to monetise all costs and benefits. The reasons for this are:

(i) that the underlying physical data do not exist. If, for example, no-one has carried out a risk assessment of, say, a given chemical, it will not be possible to say what the economic value of reducing that chemical in the environment is. The absence of basic scientific data is often a reason why monetisation cannot take place.

(ii) The underlying physical data may exist but not be in a form suitable for monetisation. Recall that the money values reflect preferences. Now suppose the physical data take the form of 'a reduction of X tonnes in biochemical oxygen demand (BOD)' in a river. Individuals do not have measurable preferences for BOD. What they have preferences for is more or less water quality. The 'object' of preferences does not correspond to the physical measure of the environmental change. This is the so-called **correspondence problem**.

(iii) The relevant physical data may exist and may correspond to what people value, but the research may simply not have been done. Consider biological diversity. There are numerous studies of the willingness to pay to conserve biological resources (e.g. endangered species) but hardly any that tells us what people's preferences are for diversity per se.

Note that we have not listed the 'impossibility' of monetisation as one of the reasons why monetisation may not exist. This is because, in principle, people have preferences for all the changes that are likely to take place in the context of policies and projects. But there will be problems if there are difficulties of perceiving the relevant change. This may be the case with very small risks, for example, or with very small changes in something like river flow.

What should be done if some costs and/or benefits cannot be monetised?

Depending on the context, it may still be possible to reach a conclusion about the outcome of the CBA. Consider the matrix below:

	Bnm > 0	Bnm < 0 ➔ Cnm > 0
Bm > Cm	Accept	?
Bm < Cm	?	Reject

Bm refers to monetised benefits and Bnm refers to non-monetised benefits. Similarly with costs, C.

Suppose monetised benefits exceed monetised costs and that Bnm > 0. Then the project or policy should be accepted even though we do not know the size of non-monetary benefits because the non-monetary benefits will simply be additional to the monetised benefits which in turn already justify the project.

If, on the other hand, Bm > Cm but Bnm is negative, i.e. there are non-monetised costs, we will not know whether to accept or reject the project/policy. But we can 'invert' the analysis and ask whether, judgementally, we think the net Bm (Bm-Cm) is sufficient to compensate for the non-monetary costs. At the very least, the procedure forces the decision-maker to list costs and benefits and to ask searching questions about the non-monetised costs.

The same procedure can be followed for the final row of the matrix.

What should be avoided is the view that if we cannot monetise everything, nothing should be monetised. This view amounts to rejecting valuable information about people's preferences. Moreover, as we see later, rejecting monetisation simply raises all kinds of other problems.

6. The issue of geographical bounds

One issue highlighted by the steering group for this paper is the geographical bounds of CBA. It is usual to set the nation as the geographical boundary, so that any costs and benefits that accrue outside the nation do not 'count' in the CBA. But the principles of CBA do not in fact favour this popular rule, for CBA is indifferent to whom the beneficiaries and losers are. National boundaries are essentially political constructs, whereas benefits and costs relate to anyone securing a welfare gain or loss.

There are in fact some good reasons for keeping the nation as the 'working rule' for the geographical bound of CBA but relaxing that working rule in certain contexts.

First, where the issue in question relates to impacts which are the subject of a binding (voluntary or legal) agreement internationally, then the relevant costs and benefits should be defined in terms of the signatories to that agreement. In the context of global warming or stratospheric ozone depletion, for example, the relevant benefits and costs would be global. If the Netherlands was deciding on its climate policies then it would be proper that any costs incurred by other countries because of Dutch policy should be presented as negative benefits of the policy. The benefits to other countries of reducing greenhouse gases in the Netherlands should similarly be included in the benefit estimates. While there may be circumstances where the CBA might be defined in terms of national gains and losses, the fact of signing an international agreement is an acknowledgement that others' gains and losses are just as relevant as domestic gains and losses.

Second, the same principle applies where the agreement is international but not global. Acid rain would be an example. The damages due to the emissions from the Netherlands that are relevant are those to the entire UNECE region because the Netherlands is a signatory to the Long-Range Transboundary Air Pollution Convention.

Third, some objects of policy may not be the subject of national property rights only. A cultural asset that is a World Heritage Site, for example, is, effectively, an asset over which the world as a whole has property rights in some form. It is then wholly appropriate to measure the benefits of conserving that site so as to include the benefits to people outside national boundaries.

Fourth, even without international legal agreements, any nation may feel a moral obligation to account for its impacts on others.

What should be done when there is no moral obligation and no legal agreement but transboundary benefits and costs occur anyway? There is a probable asymmetry here. For costs it is easy to see that there may still be a case for counting them in to the CBA. The reason for this might be a moral obligation (above) but it may also be self-interest: neglect of the costs to others now might invite similar neglect of costs imposed by another country on the Netherlands in the future. Simple self-interest dictates the ‘good deed’ approach. It is less obvious when it comes to transboundary benefits. The argument could be that it is best to include them on the grounds that this would encourage others to do likewise for reciprocal projects and policies, encouraging them to increase the size of such beneficial projects and policies. But there may be taxpayer resentment at the idea that national taxpayers are paying from non-nationals to gain from a project. The issue could be resolved, of course, if such transboundary benefits could be appropriated by pricing policies, i.e. charging others.

Overall, the geographical bounds issue is one that has probably not been treated in a very logical manner in CBA in the past. We have suggested some rules to deal with the different contexts in which this issue might arise.

7. Experience with CBA

CBA is quite widely used in some countries and hardly at all in others. Does individual country experience explain how to get CBA accepted for decision-making? It certainly helps to explain why it secures greater acceptability in some contexts rather than others.

There are two main contexts for monetisation of expressing costs and benefits in monetary terms: **regulation** and **liability**. There are other contexts, such as green national accounting and demonstrating priorities for policy, but the history of monetisation is, by and large, a history of regulation and liability. In the regulation context, it is CBA that is relevant. In the liability context, only damages are relevant so money values of damages are the relevant measure, not CBA.

Regulation
The regulatory role of CBA is most advanced in the USA. The major piece of legislation in this respect was Executive Order 12291 (1981) which required a benefit-cost assessment of new regulations which impose significant costs or economic impacts. EO 12991 required that, for any new regulation, 'the potential benefits outweigh the costs' and that 'of all the alternative approaches to the given regulatory objective, the proposed action will maximise net benefits to society'. EO 12291 helped to engrain cost-benefit thinking in federal agencies, although actual cost-benefit studies were applied in a non-uniform manner across agencies. The US Environmental Protection Agency was one of the agencies to take the requirement to carry out cost-benefit studies most seriously. But the requirements for using CBA vary across statutes.

Several court cases have established that CBA cannot be used by agencies unless explicitly authorised by statute. However, even where analysis of costs and benefits was not explicitly required, the US EPA tended to adopt regulations on the basis of CBA studies. Thus, compared to Europe, CBA is far more influential in the USA than a simple comparison of formal requirements would suggest.

Whether CBA is actually used more than the statutes require, it remains the case that US legislators quite clearly regard CBA as not being relevant in a number of regulatory contexts. It is tempting to think that this has something to do with doubts about the credibility of *benefit* estimates, but it is significant that, while the *costs* of regulation are given more consideration than the benefits, several statutes and corresponding court cases specifically exclude even the costs from consideration in standard setting. This suggests that EO12991 was never consistent with the philosophy underlying much of the environmental legislation. Nonetheless, new regulations require the approval of the Office of Manpower and Budget (OMB) and OMB could therefore enforce EO12991 over and above actual statute requirements.

EO 12991 was superseded by EO 12866 in 1993. This replaced the 'benefits outweigh costs' provision with 'benefits justify costs'. Benefits include 'economic, environmental, public health and safety, other advantages, distributive impacts and equity' and may not all be quantified. In effect there was no formal requirement that benefits actually exceed costs in a quantitative sense. Some commentators suggest that EO 12866 endorses CBA as an 'accounting framework' rather than an 'optimising tool'.

The situation with respect to CBA in the USA is therefore an uneasy one, but, in so far as the Executive Orders can be used to 'impose' CBA, CBA remains very influential in the US system. That a number of the statutes to which the EOs might apply explicitly exclude CBA considerations, and even cost considerations, suggests, however, that alternative philosophies are still important. Morgenstern (1996)[22] suggests that the whole culture of EPA is rooted in legal and scientific objectives, and some of the 1970s ideology about the 'environmental cause'. As such, economic considerations were always likely to be secondary. Moreover, economists are very much in the minority at EPA, and the educational background of most of the employees is such as to reinforce the view that economic analysis is a secondary consideration in environmental protection. Morgenstern (1996) suggests two further reasons for the 'downgrading' of economic influence in US regulation. First, there has been the rise of a 'win win' philosophy which suggests that environmental protection can be secured at no discernible economic cost. We noted earlier that this is an illusory approach to decision-making. Secondly, 'spin' and 'PR' have tended to replace awkward decision-making. This, we might observe, is not a trend confined to the USA.

In the European context, CBA has secured an increasing role in decision-making. Article 130r(3) of the Treaty of European Union (the Maastricht Treaty) requires action to take into account several factors of which one is 'the potential benefits and costs of action or lack of action'. Clearly, *some*

[22] Morgenstern, R. (1996) *Economic Analyses at EPA: Assessing Regulatory Impact*, Washington DC: Resources for the Future.

form of CBA is mandated by this provision. Up to about 1990 EU environmental policy was, by and large, effected with little formal evaluation of policy taking place. In some cases, virtually no assessment of benefits and costs was undertaken, even at the qualitative level. This appears to be the case even for major and expensive Directives such as the Wastewater Directive, Drinking Water Directive, Habitats Directive and the Bathing Waters Directive.

Some sort of formal appraisal was undertaken on water pollution and on the impacts of the Single Market before 1990. The number of studies indicated is deceptive, however, since they are often studies repeated for separate countries, or, in the case of water pollution, for different substances. Nonetheless, the fact that impacts were measured for individual Member States is an indicator of the importance of the issue. The multiplicity of studies on water tends to reflect the significant number of key pollutants involved in water pollution. The other major targeted area for simulations of costs and effectiveness was the carbon-energy tax. In turn, this proposed Directive attracted probably the single largest opposing lobby of any Commission proposal, so that the studies are a natural outcome, anticipated or ex post, of that process. Studies of Directives *per se* remain very few and these tend to be concentrated into the period since 1994.

Within individual European member States, CBA has varying degrees of influence. In the UK, for example, it has effectively become embodied in 'regulatory impact assessments' (RIAs), although the quality of those assessments varies substantially. In Norway, monetised health damages have been used extensively to guide policy on air pollution control.

Liability and damages
The second major area for the use of monetised estimates of environmental damage has been in the context of liability. In the USA this has related mainly to contaminated land and oil spills. Outside of the USA, however, there is little experience of this context for monetisation. This reflects the fact that liability legislation is less extensive, although there are examples of monetised damages being used in cases of oil spills. Additionally, damage estimates have been used in some quasi-judicial contexts such as the licensing of abstractions from rivers in the UK.

Factors explaining different uses of monetisation
What can be learned from the regulatory and liability contexts for monetisation? The essential lessons appear to be as follows:

(a) Where there is a strong tradition of 'efficiency in government', as in the USA where the tradition dates back to the pre-war era, there is likely to be a greater use of CBA.

(b) Efficiency in government as a philosophy is partly a political and cultural issue. Political parties strongly dedicated to free market principles and concerns for the 'rights' of taxpayers are perhaps more likely to believe in CBA in the belief that it will limit, rather than enhance, regulation. The fact that CBA often shows environmental regulation to be strongly beneficial may perhaps account for some of the retreat from CBA in the USA where free market principles are strong. In other words, CBA may not have had the result the free marketers thought it would have. On the other hand, CBA can, in some contexts, slow down the regulatory process by adding to the burdens of the regulatory agency.

(c) The USA has very strong lobbying groups in favour of CBA. In turn those lobbying groups may themselves have free market principles which produce an anti-regulation philosophy. An example would be the American Enterprise Institute, which is a free-market organisation that produces high quality critiques of government policy.

(d) As opposed to lobbying, the US is strong on policy analysis institutes, 'think tanks' that have the explicit purpose of monitoring and appraising government policy. It is much harder to identify comparable policy analysis institutes in Europe. Those that exist do not produce research of the quality of, say, Resources for the Future, of the World Resources Institute. This means that, unlike the USA, those within governments in Europe have few external alliances that they can make to further RDM.

(e) There is a strong tradition in some countries of secondment of academics to government, or close links between academia and government. The Central Bureau of Statistics in Norway produces not only statistical and other support services for government but is an acknowledged source of strong economic research. In the USA, staff at policy analysis institutes are regularly seconded to government. Europe has perhaps less of a tradition in this respect, although Italy is notable for the role that academics have played in formulating government environmental policy.

(f) Liability legislation is a powerful impetus to the use of monetisation. One way or the other, damages must be estimated and once liability extends to environmental damage it is difficult to see what criteria could be used for assessing damages other than monetisation methodologies of the kind used in environmental economics. The other feature of liability is that it forces monetisation 'into court' so that monetary estimates of damage have to be credible because they will be debated and disputed by differing experts.

Clearly, then, some legal or quasi-legal requirement that costs and benefits be considered in (a) **regulation** and (b) **damage liability** could open the way for wider and better use of CBA in Europe. The issue becomes one of how to persuade the decision-makers that this would be a sensible thing to do. The lessons from history are essentially that *the most powerful argument in favour of doing this is that CBA encourages the search for more efficient regulation and that it is a useful barrier to over-regulation*.

8. Obstacles to the use of Cost-Benefit Analysis

The philosophy of CBA gives clues to why it is sometimes difficult to secure adoption of CBA in practice. We classify the obstacles as issues of **philosophy and content**, and as issues of **process**.

Philosophy and content

1 **Credibility** – CBA is a quantitative technique and the resulting quantities are often **uncertain**. Decision-makers will generally be averse to uncertainty.

2 **Morality –** CBA uses **money values** and there is often a ‘moral’ hostility to using money as the measuring rod.

3 **The efficiency focus**– CBA has economic efficiency as its goal. But governments have multiple objectives, hence CBA appears to be partial and non-comprehensive.

4 **The democratic principle** – while it may seem odd to suggest that decision-makers oppose ‘democracy’, there are concerns about the legitimacy of reflecting preferences in all contexts.

Process

5 **Flexibility** – some decision-makers may feel that CBA compromises their flexibility of decision-making.

6 **Participation** – CBA is sometimes criticised for being non-participatory.

7 **Capacity** – CBA requires a certain level of expertise on the part of those using it or judging its results.

We address each of these issues in turn, and in doing so, suggest ways in which the obstacles may be reduced. Before doing that, we need to address a very important issue that is rarely addressed in debates about the 'proper' way to make decisions. This is the **baseline issue**. This issue concerns what we would do if we did not adopt CBA as the general basis for making decisions.

9. Baseline

CBA is a coherent decision-making guide based on the concept of economic efficiency, as noted above. But it is only one such guide. If decisions are not made with the help of CBA then they must either be made on an **ad hoc basis**, i.e. with no particular systematic guidance, or using **some other guidance procedure**. On the assumption that purely ad hoc decision-making is rejected on the grounds that it is not rational, the alternatives to CBA are the other systematic procedures that might be used. While this may seem obvious, one of the curious features of the debate about CBA is that it is often presented as if the alternatives to CBA are somehow free of the criticisms that are advanced against CBA. This is the 'baseline' issue. If CBA is challenged, it is logically proper that the critic offers an alternative procedure which is 'better'.

The very first test of the alternative approach is to see if it is free from the criticisms that have been advanced against CBA. Establishing this rule is essential in the discourse about RDM.

Space forbids that we discuss the alternative procedures in any detail (see EFTEC, 2001)[23]. The main 'contenders' for rational decision making are:

Environmental impact assessment (EIA)
Life cycle analysis (LCA)
Risk assessment (RA)
Multi-criteria analysis (MCA)
Cost effectiveness analysis (CEA)
Cost-benefit analysis (CBA)

Table 9.1 below shows that techniques are often complementary. CBA cannot be carried out, for example, in the absence of risk assessments of health impacts or of environmental impact assessment in some form. Similarly, LCA can be combined with CBA to produce combined economic valuation and life cycle assessment. Similarly, LCA and EIA have close similarities if the EIA takes on a cradle-to-grave impact stance. EIA is, of course, also essential for CBA: all valuation techniques require the physical impact data to work with. Finally, even if techniques appear to be different, they might profitably be combined. For example, a CBA might be combined with some scoring and weighting procedures (MCA) for those impacts that are not readily monetised. The interconnections between the techniques indicate that a debate about which technique is better can easily be misleading. The techniques tend to build on each other. Nonetheless, we can try to see whether often

[23] EFTEC (2001), *Guidance on Using Stated Preference Techniques for the Valuation of Non-Market Effects*, report for the UK Department of the Environment, Transport and the Regions, London (summary guidance report forthcoming at www.detr.gov.uk).

quoted criticisms of CBA are avoided by adopting one of the other techniques. Annex 1 lists each procedure and highlights the main features and the advantages and disadvantages of each one.

The overview in Annex 1 serves to reveal a number of important observations about so-called decision rules.

First, **only two appraisal techniques actually provide a decision rule**. A decision rule is one that indicates whether the policy or project is potentially ‘good’ or ‘bad’. Decision rules embody tests to determine this. **CBA** has such a test, namely that benefits exceed costs. No other appraisal technique embodies this test. Comparative Risk Assessment (see Annex 1) comes close because it has a rule that says the choice should be that option which has the highest risk reduction per Dfl expended. But it is limited to this context. Nor does it say whether *any* risk reduction option should be accepted, i.e. it answers the question A or B? but does not answer the question: A or B or neither?

Table 9.1 Relationships between evaluation techniques

	LCA	**RA**	**MCA**	**CBA**
EIA	EIA can be done on the basis of life cycle impacts. Usually not done.	RA could be an input to EIA provided EIA deals with potential damages, not just emissions.	EIA needed to provide the impact data for the MCA. Environmental impacts would be one 'goal'.	EIA essential as an input to CBA.
LCA	XXXXXXXX	Risks could be evaluated on a life cycle basis. Generally not done.	MCA could take life cycle impacts as one goal.	LCA consistent with the 'with and without' principle of CBA.
RA	XXXXXXXX	XXXXXXXX	RA essential as input to MCA.	RA is an essential input to CBA, e.g. premature mortality, morbidity etc.

Second, decision-makers may not want an explicit decision rule. As discussed later, they may prefer a decision technique which leaves them **flexibility of choice**. Several rules provide this flexibility, e.g. MCA, CEA. CBA and CRA, on the other hand, present the result in terms of a ‘cut and dried’ rule.

Third, and contrary to the way in which they are often discussed, some of the techniques are only **inputs** to decision-making. They do not provide guidance on how to make decisions. LCA and EIA fall into this category. LCA is often presented as if it provides a decision-making rule, but it only begins to do this if the life cycle **inventory** is weighted by factors reflecting the importance of the impacts. Some LCA does this, but procedures for selecting **weights** are often arbitrary, i.e. have no underlying theoretical rationale. Where the weights have a rationale, as with monetary damage weights, LCA would effectively be equivalent to CBA if other costs and benefits are also taken into account. Invariably they are not accounted for in LCA.

Fourth, few of the techniques explicitly address **time**. There are several ways in which time should enter an analysis. First, impacts are distributed over time and if the weighting procedure is preference based, future impacts should be ‘**discounted**’ to reflect the fact that individuals care less about the future than the present. The idea of discounting the future is thought by some to be inconsistent with the objective of **sustainability**. This not necessarily true. Even governments tend to discount the future. What can be said is that techniques that fail to address the discounting issue (even if they conclude that future impacts should not be discounted) must be at least non-comprehensive. The second effect of time is that some impacts will become less or more important over time. This is different to discounting and simply refers to the fact that individuals' intensity of preference for something may change, often with income. In CBA this issue is handled by a **falling (rising) relative price**. Other techniques appear not to address the issue at all. Note that CBA explicitly addresses both time-related issues[24].

The failure to address the baseline issue in public debate is not confined to the comparison of CBA with other techniques. It is just as common, for example, in discussions about market-based instruments such as environmental taxes. Taxes, it is often said, will harm the poor and do damage to a nation's competitiveness. Apart from the fact that these statements are usually made without supporting evidence, the fact is that the same criteria are not used with respect to the alternative form of regulation. Thus, it may well be the case that traditional regulation does more damage to competitiveness and to low income groups than the tax option. Instead, the arguments tend to be presented as if the alternative to taxation is no regulation at all. The phenomenon is common in politics generally (e.g. 'single issue' pressure groups).

How might the baseline syndrome be overcome? For advocates of any one procedure (e.g. CBA), it is important that they have a list of criteria by which the efficacy of a policy measure might be determined. These are not likely to be controversial (e.g. efficiency, equity, sustainability, impairment of competition etc.). The controversy arises because interest groups select only one or two of the criteria and argue about these, effectively giving the others no weight at all. *Thus, an advocate of CBA should first list the criteria, perhaps using policy statements already made, and should then ask the question: how would the alternatives to CBA compare against these criteria?* Interest groups seeking to confine the debate to one or two criteria only should be asked why they ignore the other criteria and to place themselves in the position of a decision-maker who has to listen to, and account for, different interests and stakeholders. The **criteria/alternatives matrix** is a good way of organising this debate.

Determining the baseline is absolutely critical to policy debate. All too often, debate takes place as if the alternative to any given policy guidance procedure is to do nothing, to muddle through or to use ad hoc procedures. In other debates, criticisms of one procedure, usually CBA, are stated without recognition that the same criticisms often apply to the alternative procedure, and without recognition of the fact that other procedures have other problems or that they are incomplete decision guiding procedures. This section suggests that the use of matrix approaches can help different stakeholders see the problem through the eyes of other stakeholders. At the very least, the matrix lays down the basis for rational debate.

10. Obstacles: credibility

One of the criticisms advanced against CBA is that the answers are not credible. Credibility rests on several factors: whether the final estimate of net benefits has **'too wide' a range**; whether the **assumptions** made in order to achieve the estimates of net benefits are themselves credible; and whether the estimates have been truly **tested for their validity**.

All are valid sources of concern in decision-making. The range of estimates tends to represent the underlying **uncertainty** of socio-economic data. Social science data are not like 'physical' data. While both are subject to uncertainty, social science data are far more probabilistic since they reflect the behaviour of millions of individuals. Thus uncertainty is endemic to social science. *In terms of choosing between decision-making guidance, however, uncertainty per se is not the issue. What matters is whether any one form of guidance is more uncertain than the others*. Annex 1 shows that most guidance procedures are incomplete in that they do not account for all the information that one would expect to be included in a rational decision. Thus, LCA and EIA make no reference to the costs of policies or projects, yet no rational decision can be made independently of costs for the reasons raised in Section 1 above. Hence, even if CBA is more uncertain than other techniques (an assumption

[24] But the state of play with respect to how these issues are addressed is not satisfactory. There is no consensus on appropriate discount rates nor is there much evidence on the correct rate at which relative prices change.

that needs to be tested), it does not follow that some alternative technique is better. The greater uncertainty of CBA is simply being exchanged for a somewhat illusory certainty, illusory because it is achieved by simply ignoring other factors that should bear on how to make decisions. Consider CBA 'versus' risk assessment. CBA may appear to be more uncertain than Risk Assessment because the money values appear to add to the variables that are uncertain. Risk Assessment avoids monetary assessment and hence reduces the level of uncertainty. But it does so by sacrificing a basic requirement of decision-rules, namely telling us whether something is good or bad, desirable or undesirable. In effect, with Risk Assessment we have no idea whether any decision is 'correct' because we have no absolute standard against which good or bad is measured. Risk Assessment does tell us that policy A is to be preferred to policy B if A secures more risk reduction per Dfl expended compared to B. But it does not tell us that A is itself desirable. Hence a whole different layer of uncertainty is introduced with Risk Assessment.

The first rule for dealing with the 'uncertainty means non-credibility' argument is therefore to raise the baseline issue again. What are the sources of uncertainty for each procedure? If CBA has one or more sources of uncertainty, are these sources absent in the alternative procedures? What other forms of uncertainty do the alternative procedures have? Sometimes this procedure is known as '**the best game in town**' argument: all approaches are imperfect – the issue is one of choosing the least imperfect.

But CBA may have some **special sources of uncertainty**. First, it assumes that individuals themselves behave rationally[25]. The rationality assumptions underlying CBA are quite strict. For example, it is assumed that individuals are capable of consistent preferences across a wide range of goods and services, including non-marketed environmental goods. Psychologists suggest that individuals may not in fact behave this way. They may, for example, break rules of '**transitivity**', effectively preferring A to B, B to C but C to A. In other respects they may 'trade' between options within a given '**mental account**' but not between mental accounts. A mental account might be, say, the weekly shopping budget, while housing expenditures are in another account, and entertainment in yet another. They may have rational preferences within any account but not be willing to trade between, say, housing and shopping. Intransivity and mental accounts both create problems for the assumption that people have clearly defined preferences across all goods and services. Again, it is important to apply the baseline test. Are these problems avoided by adopting a different procedure? The answer is that, as long as the procedure is rooted in the idea that people's preferences 'should count', these problems are not avoided. They are, of course, avoided if people's preferences are rejected as the basis for decision-guidance. Indeed, it could be argued that the fact of imperfect rationality is a ground for rejecting individual preferences. As noted in Section 1, ultimately, different value systems cannot be avoided, but it was suggested there that the overthrow of individual preferences is something that should be avoided unless absolutely necessary.

A second source of uncertainty of relevance to CBA is that pertaining to **stated preference techniques**[26]. A stated preference technique is based on a questionnaire of a random sample of the population. Individuals are asked about their attitudes generally, about their attitudes to the specific good, and about their socio-economic status. In **choice modelling** procedures they are also asked to rank or rate a given package of attributes that together make up a project or policy, relative to some other package. They are not asked their explicit willingness to pay, but a price or cost of some sort is included as an attribute in the packages[27]. The analyst then infers the WTP from the answers provided. In **contingent valuation** the respondent is asked directly for his/her WTP, sometimes in response to

[25] More strictly, that they behave rationally as a group, i.e. 'on average'.

[26] EFTEC (2001), *Guidance on Using Stated Preference Techniques for the Valuation of Non-Market Effects*, report for the UK Department of the Environment, Transport and the Regions, London (summary guidance report forthcoming at www.detr.gov.uk).

[27] Terminology varies, but choice modelling techniques include choice experiments, contingent ranking, contingent rating and pairwise comparisons. These are sometimes referred to as conjoint analysis. Note that not all variationsof choice modelling are equally robust.

an open-ended question (what is your maximum WTP?) and sometimes in response to a close-ended question (are you willing to pay DflX?)[28]. These approaches, which are increasingly used in cost-benefit studies, are thought by some to *add* to the uncertainty in CBA. The reason for this is that the questionnaires are hypothetical and hence the answers are hypothetical. The hypothetical nature of the questionnaire is not itself a criticism. After all, the reason hypothetical questions are being asked is invariably because there are no 'real' markets for the analyst to refer to. If the real markets existed, we would not need to ask hypothetical questions. Nonetheless the answers could be biased (upwards or downwards, but more usually it is thought the bias is upwards). The issue becomes one of finding out how likely it is that the hypothetical answers diverge from the respondents' 'true' WTP. To this end, stated preference techniques adopt many tests of validity. A well-designed and implemented questionnaire should therefore minimise the biases. An example would be to conduct a stated preference study at the same time as one using some other technique. An example might be a hedonic price study whereby the influence of the environmental change in question on some market price (e.g. house prices) is determined. The WTP 'revealed' through the hedonic technique can then be compared with the WTP derived from the stated preference technique. But bias is likely to remain. This may not matter too much if there is some idea of the direction of bias and its probable scale. Research suggests that there is an upward bias in WTP responses, but it is not easy to say what the scale of this bias is.

While critics of stated preference techniques are often unaware of the tests embodied in a good study, the feeling that answers are not reliable indicators of true preferences remains. Since stated preference analysis is an ongoing activity, and a great deal more research needs to be done, it may be that future research will resolve the issue of how large the hypothetical bias is. Accepting that, for the moment anyway, we do not know, the same baseline issue arises. If we adopted any other technique, would we avoid the supposed additional bias in stated preference answers? One view suggests that we would not avoid this bias because it is a feature of the hypothetical nature of the questionnaire. The bias would therefore also be present in any technique that requires the views of potential beneficiaries and losers to be determined. On another view, it is the very fact that money is involved that produces the bias, or at least part of it. If so, perhaps choice modelling techniques where money is not explicitly mentioned would avoid the problem. At the moment, we do not know if this is the case because so few choice modelling experiments exist (outside of the transport field anyway).

Ultimately, then, the issue of uncertainty reduces to:

(a) determining whether it is any better or worse if some other technique is used; and
(b) the view taken about the 'special' bias that may reside in the use of stated preference techniques. Note, of course, that most cost-benefit studies still do not make use of stated preference techniques so that, even if it is felt there is some special bias, this does not mean that most CBA is biased.

11. Obstacles: moral objections to Cost-Benefit Analysis and the issue of democracy

Some critics of economic valuation and CBA find its emphasis on opportunity cost morally objectionable. It is often argued, especially in the health care context, but also in safety and environmental contexts, that certain outcomes of policies should not be subject to budget considerations. Patients, it is said, should be treated with the best health care available regardless of cost; road and rail safety should be an absolute priority regardless of cost; and environmental protection is an absolute moral imperative and cannot be subject to rationing by cost. The main difficulty with these views is that they ignore the meaning of cost. As Section 1 noted, the proper measure of cost is the benefit that is forgone by allocating expenditure to a chosen project or policy. If money is spent on A it cannot be spent on B. While A may have some of the characteristics of

[28] Close ended question formats take the form of bidding games, dichotomous choice and payment ladders.

being 'moral goods', like health and safety, so might B. Money spent on road safety cannot be spent on cancer research or hospice care. The principle can be extended. If funds are raised to finance a risk reducing expenditure, the effect may be to impose risks on the people who are taxed to raise the funds[29]. The essential point is that cost is not 'just money': it is an expression of resources that could be used for all kinds of other, perhaps equally deserving, purposes.

More fundamentally, some critics object that, by focusing on the preferences *of individuals*, economic evaluation takes account only of **self-interest**. If an individual has a preference for or against something, it might appear that the preference will be formed on the basis of what that individual judges to be best for himself. A short way of expressing this point is to say that the individual acts out of self-interest. Indeed, this is how '**consumer sovereignty**' or '**economic rationality**' is often characterised. But the issues for which monetary valuation will be used will often be those where the **public interest** is the issue, i.e. what is best for society as a whole.

Whether or not the public interest is the same thing as the sum of individuals' self-interests is a controversial question in political philosophy. So too is the question of whether it is better if, when acting in the political arena, individuals act in pursuit of their private interests, constrained only by rules of procedural fairness, or if they act on the basis of 'public spirited' or 'citizens' preferences. Viewed in some philosophical perspectives, it is wrong to reach collective decisions by adding up self-interested preferences. The proper procedure is to ensure that the context of decision-making is one where citizens' preferences are expressed. That context would appear to be the political arena, not the outcome of a context where, say, experts collect questionnaires from respondents who are asked for their stated preferences[30]. (We say 'appear' because, it is far from clear that preferences revealed in the political process are less self-interested and more public-spirited than those revealed by the same people in the market.)

As a first response to this criticism, we emphasise that CBA is not a substitute for the political process; it merely provides information to the actors in that process. Someone who believes that a particular policy is morally required, or is morally prohibited, can properly try to persuade other political actors to share this belief and to act on it, whatever the results of a CBA. For example, if someone believes that everyone has a moral obligation to accept a certain increase in taxation in order to pay for a programme to conserve endangered species, the fact that the programme fails to satisfy the CBA test does not require them to change their belief: they are entitled to conclude that other people are evading their moral obligations. But the CBA results remain meaningful information about what people really are willing to pay for the programme.

However, it is a mistake to think that CBA takes account only of self-interested preferences. It takes account of whatever preferences people have, for whatever reasons they have them. For example, a person may want an environmental asset to continue in existence even though he makes no direct use of it, nor intends ever to make any use of it. To the extent that he is willing to forgo other things that he values in order to conserve the asset, he has a preference for its conservation and its conservation is a benefit to him. He might be motivated by fact that without the asset's continued existence, he could not enjoy it vicariously through television or film. This motive might be described as self-interested. But he might also be motivated simply by wanting the asset to continue in existence for its own sake, or because he sees himself as a 'steward' of the environment, or because he wants it for his children or future generations to enjoy. In short, *there is nothing in the concept of preference that tells us that preferences have to be motivated by self-interest.*

Overall, then, those who hold moral views about CBA may well be objecting to the underlying utilitarian philosophy of CBA. But they may also share some confusions about the nature of economic

[29] This aspect is explored in the literature on 'risk-risk' analysis. See W K Viscusi (1998) *Rational Risk Policy*, Oxford: the Clarendon Press.

[30] This is the view taken, for example, by Mark Sagoff. See M Sagoff (1988), *The Economy of the Earth: Philosophy, Law and the Environment*, Cambridge: Cambridge University Press.

valuation and the fundamental role played by the concept of opportunity cost. How might these confusions be reduced? Again, there is no substitute for reasoned discourse. The CBA advocate should always be sure that those objecting to it are asked whether the moral implications of their views have been fully worked out, by seeking to explore the cost of those views and the implications of ignoring that cost. Moral objectors may then still have a residual objection to CBA based on the view that balancing gains and losses ignores the 'absolute rightness' of avoiding a particular form of loss. The political process, if fairly formulated, is then the right place to debate those claims. What is not justified is excluding consideration of costs and benefits from that process.

12. Obstacles: the efficiency focus of Cost-Benefit Analysis

Section 1 noted that governments have **multiple objectives** and those objectives often conflict**.** One procedure for 'trading off' objectives is **multi-criteria analysis** (MCA). MCA does not require the monetisation of objectives but it does require a rational analysis of what has to be surrendered for what. MCA is discussed further in Annex 1. The important feature of MCA is that it embraces objectives that CBA appears not to embrace. For example, it could include a distributional objective (fairness, equity), some assessment of sustainability, and wider national concerns such as competitiveness, employment, regional balance etc. The matrices introduced in Section 2 were in fact the kind of matrix with which MCA would begin. The advantage of these matrices is that decision-makers are forced to set out the criteria by which policy might be judged, and they are forced to quantify the effects of policy choices. Some policy analysts draw attention to another important feature of such matrices: they make it clear to everyone else, the public included, what the analytical reasoning is behind potential decisions[31]. This encourages openness and appreciation that disciplines other than economics have valuable inputs to make.

If MCA is 'wider' than CBA why not recommend MCA rather than CBA? The question is somewhat misleading because CBA is in fact a particular form of MCA. There is nothing in MCA that says efficiency is not important and nothing that says impacts should not be monetised where appropriate. If there is an 'equity' goal this may not be suited to monetisation, in which case something that is more efficient but less equitable must be traded against something that is less efficient and more equitable. The efficiency status cannot be determined, however, without some form of CBA[32]. Thus CBA can, and should, be an input into MCA.

Proceeding to MCA without CBA can also be dangerous. Some of the problems are:

(i) Many MCAs do not account for public preferences at all, but use expert judgement. This runs counter to the 'democratic' principle already introduced;

(ii) MCAs face considerable difficulties with time (discounting and changes in relative values); and

(iii) MCAs often risk double counting of objectives.

Provided great care is taken over the preparation of MCA, then, it is, as interpreted here, a wholly rational way of presenting policy options. But it in no way precludes CBA which must be part of it.

[31] This is stressed, for example, in D MacRae and D Whittington (1997), *Expert Advice for Policy Choice*, Georgetown University Press, Washington DC.
[32] CBA can in principle integrate equity and sustainability into its analysis. Few CBAs attempt to incorporate sustainability requirements, but integrating equity concerns was once very popular in CBA practice.

13. Obstacles: flexibility of process

An important institutional factor that inhibits the use of CBA is the demand by policy makers for **flexibility of choice**. Imagine a technique, like CBA, say, that produced precise estimates of costs and benefits such that the benefit-cost rule could be clearly and immediately identified. If CBA were the *only* form of policy guidance, the policy-maker would merely have to assent to whatever the CBA result produced. This limits the freedom of choice of the policy-maker and hence makes him or her resent techniques that appear to usurp that freedom. Decisions become 'technical' rather than a matter of judgement. *The more quantitative the technique, and the more it embodies a decision rule in the sense advanced earlier, the less flexibility that approach will appear to afford the decision maker.*

Flexibility is a legitimate concern, but it can be preserved whilst still employing CBA. No one technique should appear as providing the 'right' solution. It is an added piece of information which at least orders the information available in some logical way. Thus, information on what the likely impacts are of a policy is the least that should be expected. How far quantification and aggregation of those impacts then take place depends on the context. But, there has to be some guidance. It is a matter of 'appraisal thinking' rather than the provision of exact numbers and outcomes. Again, the **alternatives/criteria matrix** helps because it places the CBA result in a wider context. The decision-maker is then free to introduce other criteria and to weight them accordingly.

The matrix/CBA approach is to be preferred to one alternative widely met in decision-making contexts. Many would argue that what matters most is **how decisions are made**. It may be that a procedure using public participation is regarded as more important than one based on some balancing of costs and benefits. In some cases, the two may be combined, as with modern approaches to economic evaluation through the adoption of questionnaire-based approaches. Process is also important at the political level: consultation with parties who are, or represent in some way, the stakeholders offers some guarantee that the interlinkages referred to in previous chapters will be captured. But **process is not sufficient**. Processes have to be informed by facts and by some ordering of the facts. Essentially, all the appraisal techniques reviewed in this chapter are about this process of ordering, of placing gains and losses into some framework so that they can be compared. Another way of thinking about it is that **process does not guarantee rationality of decisions**. Equally, decisions cannot be made and cannot be effective without efficient process, not least because those excluded from the process may well 'block' a decision or at least make its implementation difficult.

14. Obstacles: is Cost-Benefit Analysis non-participatory?

Modern approaches to project and policy appraisal rightly stress the need for **public participation** in the process of appraisal. While participation is often seen as an end in itself, it is also a **necessary ingredient for economic efficiency**. The reason for this is that lack of participation can easily engender opposition to a project or policy, making it difficult to implement and costly to reverse. Participation may also produce better policy and project design since those most affected are closer to the issue than analysts and decision-makers. In the economic development literature, it is well established that development projects are more likely to succeed if communities and gender groups are involved in the process. Appraisal techniques are often criticised because they may omit this participatory feature of decision-making. **Stated preference** techniques, however, have an important role to play in securing participation, a role that emanates directly from the fact that the techniques elicit all kinds of information about attitudes, motivations, preferences and willingness to pay/accept compensation.

The literature is not always clear on the meaning of the term 'participation'. At least three versions of the term appear: (a) participation as consultation, i.e. taking account of the preferences of affected parties; (b) participation as influence, i.e. ensuring that affected parties influence the direction and form of the project or policy; and (c) participation as benefit-sharing, i.e. ensuring that affected parties receive a share of the resulting benefits. In any of these contexts great care needs to be taken that all

genuine stakeholders are consulted. Stakeholders may include non-users, so that limiting the focus of participatory processes to directly affected parties only can be inefficient. This reinforces the need to ensure that **proper sampling of opinion** is undertaken. Simply working with a **focus group** of a handful of people is unlikely to meet the requirement that a reasonable sample of opinion be surveyed. Even in the context of wider ‘community meetings’, participants may be far from being a random sample. Meetings may be dominated by activists, by those who can afford the time to attend such meetings, and may be subject to strategic behaviour because of the potential for any one individual deliberately to influence what others say. The more discursive approach to participation can, however, enhance the amount and perhaps the quality of information provided because responses tend to be more open-ended and not restricted to the questions that the analyst tends to ask.

CBA can therefore be participatory if it makes used of stated preference techniques. Moreover, there are good reasons to suppose that the rigorous statistical requirements for sampling will make a stated preference approach more useful for participatory analysis than other, more conventional approaches to participation.

15. Obstacles: capacity

A final obstacle to the use of CBA is the fact that, like all economic techniques, it requires an input of time in order to understand the underlying rationale and some of the technical details. However well trained decision-makers are, there will always be a residual element that does not invest time in trying to appreciate the logic of CBA. Much the same goes for other guidance techniques and, for that matter, with scientific analysis generally. There are no easy cures here. Decision-makers are human beings, given to all the foibles, prejudices and irrationalities that make humans generally rather interesting. The matrix approach suggested in this paper would, however, help to expose those who would want to reject rational criteria and who would refuse to measure gains and losses at all.

16. Getting Cost-Benefit Analysis into the process of decision-making

Clues about the ways in which CBA can be introduced to the decision-making process have been set out previously. They are gathered together in this section, together with several other observations.

First, there must be a rational discourse about decision-making in general. We suggested that this should be based on something like the alternatives-criteria matrix, which then forces the issue of trade-offs to be recognised. Once this issue is understood, it is a small step to choosing between the very limited number of RDM techniques that address the issue of weights and objectives in a comprehensive manner.

Second, CBA is unlikely to be brought into the process if it depends solely on the initiatives of groups within government. There must be external support in the forms of academic or research institutes, or NGOs, who favour the procedures. In turn, those institutes provide a forum for further debate, for advancing the theory and practice of CBA, and they can act as peer reviewers and independent assessors. In short, there is a need for alliances between groups in government and groups outside, whilst at least some groups outside must retain their independence.

Third, there must be an ‘efficiency culture’ for CBA to thrive. If public spending is seen as more of a morally driven issue than one of efficiency, the chances that CBA will succeed are smaller. That said, it is incumbent on those who see spending as a moral issue to understand and account for opportunity costs, i.e. for the moral consequences of inefficient spending.

Fourth, there needs to be some tradition of ‘senior ministries’ such that a ministry of finance is regarded as the ultimate guide on what is and what is not the proper subject of public expenditure. In the UK, no decisions about taxes or public expenditure can be made without Treasury approval. It is significant, then, that it is the Treasury that issues guidelines on how to appraise policies and projects, guidelines which broadly embrace the principles of CBA.

Fifth, CBA can appear as a dry, technical subject which deters close scrutiny and invites quick criticism based on limited understanding. It is important that the subject be communicated in as comprehensible manner as possible. Academics are not always keen to popularise their subject since their own reward systems do not encourage this. Yet popularisation is important since that is the only way that the media will take an interest in the issue.

The suggestion in this paper is that CBA is best seen first as a way of thinking and organising information; second as a significant part of an alternatives/criteria matrix approach to decision-making; and as a way of introducing a rational balancing of gains and losses in a world where trade-offs are the norm, not the exception. Thereafter it is an issue of forming alliances and arguing the case.

Appendix A Types of formal appraisal procedures

Table A.1 Environmental impact assessment (EIA)

Lists and measures in physical terms the various environmental impacts from a policy or project.	
Advantages	**Disadvantages**
Forces systematic consideration of environmental consequences of actions, as input to decision-making process. Needed for techniques that do have a decision rule. Points way to measures needed to mitigate serious negative impacts	Does not provide decision rule. Does not aggregate environmental impacts. Does not have conceptual link to non-environmental impacts. No obvious way of treating time dimension.

Table A.2 EIA plus scoring approaches

Lists and measures in physical terms the various environmental impacts from a policy or project. Aggregates impacts using expert 'scores' and 'weights'.	
Advantages	**Disadvantages**
Potential comprehensiveness high as any given impact can be included. Easy to use.	Weights and scores may be arbitrary. Reduces to CEA if weighted scores taken as measure of effectiveness. No obvious link to non-environmental impacts. No clear decision rule. No obvious way of treating time. Experts are knowledgeable but unlikely to be representative.

Table A.3 EIA plus distance-to-goal methods

EIA with weights assigned to impacts according to the size of the gap between status quo and intended policy targets.	
Advantages	**Disadvantages**
Easy to understand. Most impacts can be included and considered under this approach.	Policy targets may not be the 'correct' ones from either a scientific or social viewpoint. Reduces to CEA if the weighed 'distances' are seen as a measure of effectiveness. No clear decision rule. Implicitly assumes all targets are equally important unless weights are attached to impacts. Below the target level effects are assumed to be absent or not worthy of consideration. No obvious treatment of time.

Table A.4 EIA plus economic control costs

EIA with weights determined by the cost of achieving pre-specified targets.	
Advantages	**Disadvantages**
Easy to understand. Can be extended to most impacts.	All the above, plus control costs are unlikely to be related to damage and effectively imply benefit-cost ratios universally equal to 1.

Table A.5 EIA plus economic damage costs

As above EIA but with weights determined by willingness to pay to avoid negative impacts.	
Advantages	**Disadvantages**
Clear methodology. Easy to use. Consistent with the goal of maximising economic well-being. Allows comparison of environmental and financial costs. Possible to consider disamenity impacts, which are excluded by most other methods.	Limited by availability of economic damage cost estimates. Damages and benefits are efficiency-related and may not embrace other policy goals (e.g. distributional incidence, sustainability). Reduces to CBA once other monetised impacts are included. Time only considered if converted to a CBA.

Table A.6 Strategic environmental assessment

More pro-active, wider ranging form of EIA and applied more at the broader policy level.	
Advantages	**Disadvantages**
Has potential to address environmental concerns at early stages of decision-making. Forces thorough consideration of cumulative impacts of several projects or policies. Forces wider consideration of alternatives. Proactive approach, with potential to steer developments to most appropriate locations from a national perspective.	No consensus on an official methodology. Has been considered more at theoretical than practical level. Difficult to define methodology which is suitable and consistent across all sectors and levels of policy. Does not appear to address issue of weighing impacts.

Table A.7 Life cycle analysis

Form of EIA in which impacts are measured in physical terms over the full life cycle of the product, policy option etc. Issues are the same as for EIA.	
Advantages	**Disadvantages**
Thorough procedure for evaluating environmental effects of a product, process or activity.	Reliability of results highly dependent on weighting procedure used to aggregate impacts. Weighting procedures often not used at all. Reduces to CBA if impacts are weighed by damage costs and non-environmental impacts allowed for. As practised, tends to ignore disamenity impacts and time.

Table A.8 Risk assessment

In its basic form, RA assesses the probability and scale of damage to human health and/or ecosystems.	
Advantages	**Disadvantages**
Requires identification of issues that do concern people: probability and scale of adverse effect.	May not give full indication of adverse effect, merely exceedance over some no-effect or 'acceptable risk' level. No explicit requirement to consider cost of reducing risk. Ignores benefits.

Table A.9 Comparative risk assessment

Compares risk reductions from different targeted issues per unit cost of risk reduction.	
Advantages	**Disadvantages**
Requires identification of probability and scale of adverse effect. Forces consideration of cost and the 'productivity' of resources in terms of risk reductions in different areas/ or ensures that tolerable risk levels are set according to what is revealed to be tolerated elsewhere.	May focus on one form of risk only (e.g. health risks), when other forms of risk, (e.g. to ecosystems), may be just as relevant. While CRA indicates which risk it is more efficient to reduce, used in isolation it does not identify whether *any* risk at all should be reduced. Equivalent to a cost effectiveness criterion.

Table A.10 Risk benefit assessment

Compares risks of an activity with the benefits of the same activity.	
Advantages	**Disadvantages**
Where it takes the form of risk-to-benefit ratios, it provides a partial decision rule. Can be equivalent to CBA if risks and benefits are monetised and other costs included. Forces consideration of the benefits of undertaking risky measures: risk alone is an incomplete decision criterion.	Monetisation of risks may not always be possible, and is sometimes not politically acceptable. In its risk-to-benefit ratio form it tends to ignore costs and other factors relevant to rational decisions.

Table A.11 Risk- risk analysis

Requires that a risk reduction measure accounts for the risks that may be imposed by responses to the measure itself, e.g. banning saccharin may increase sugar consumption.

Advantages	**Disadvantages**
Forces consideration of the behavioural response to policy measures. These may offset risk reductions from the policy in question.	It is an incomplete decision rule since it ignores cost. It may also focus exclusively on health risks, but need not do so. A regulation might pass an RRA test but not a CBA test. Once additional risks are introduced, e.g. ecosystem risks, some procedure for aggregation is required.

Table A.12 Health-health analysis

Compares risks reduced by a policy with the risks associated with the costs of the policy. Costs reduce incomes and lower income associated with higher risks.

Advantages	**Disadvantages**
Forces consideration of the true meaning of cost as foregone benefits.	A regulation might pass (fail) an health-health analysis test but fail (pass) a CBA test, e.g. lives saved may be > lives lost, but value of net lives saved < cost of policy. The focus is on health alone, but the procedure tends to focus on mortality alone, ignoring morbidity. Also health benefits arising from employment due to the policy are ignored. Slight real income changes are unlikely to result in significant reductions in risk aversion expenditures.

Table A.13 Multi-criteria analysis

Compares costs of policy to sets of 'outputs' that are not necessarily measured in money terms.

Advantages	**Disadvantages**
Enables quantitative and qualitative data to be combined. Permits explicit trade-off situations to be identified. Range of impacts wider (multiple goals). Probably cheaper than CBA and more comprehensive than RA.	No clear criteria for selecting impacts. Risk of double counting impact categories. Potential for arbitrariness in ordinal scoring of qualitative impacts. Potential for arbitrariness in weighting overall impacts for relative importance.

Table A.14 Cost effectiveness analysis

Compares to costs of measure to some indicator of effectiveness but not in monetary terms.	
Advantages	**Disadvantages**
Secures 'value for money' from a given budget. Essential requirement for any rational policy.	Does not say whether a given option is intrinsically worthwhile, merely whether the option is better than some other option. Becomes MCA if there is more than one indicator of effectiveness.

Table A.15 Cost Benefit Analysis

Advantages	**Disadvantages**
Permits determination of 'absolute' desirability of a policy in economic efficiency terms. Consistent underlying theoretical foundations. Forces consideration of cost as an indicator of foregone benefits (opportunity cost). Consistent with individuals' preferences (democratic base). Explicit treatment of risk and uncertainty via sensitivity analysis, risk-equivalence models, decision - theory etc.	Deals with economic efficiency only, or possibly with efficiency and distribution only. Tends not to accommodate other policy goals. Potential for discrimination against sustainability concerns. Data may not permit all benefits and costs to be monetised. Some public acceptability issues: some pressure groups continue to see it as 'unethical'.

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