23 Understanding and accounting for the costs of inaction

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In political decision-making processes, the burden of proof is often distributed such that policymakers only respond to early warning signals from environmental hazards once the costs of inaction have been estimated.

This chapter revisits some key environmental issues for which estimates of costs of inaction have been carefully developed over many years of research. The aim is to consider the methodological challenges involved in producing estimates that are credible and appropriate rather than present specific estimates for these costs.

The case studies also provide insights into how early warning signals might provide a basis for estimating the costs of inaction, when the science base is less consolidated. For example, the case of nitrates in drinking water illustrates that a precautionary approach to the costs of inaction is quite conceivable. The phase-out of ozone-depleting substances, where early-warning scientists successfully alerted the world to the damaging effects of chlorofluorocarbons (CFCs), provides another important case because additional impacts for global warming actually cause the costs of inaction to be considerably higher than initially believed. This is a reminder that figures for the costs of inaction have often been grossly underestimated.

Finally, in the case of air pollution, making use of different estimates for mortality risk avoidance will help decision-makers to see that there are higher- and lower-bound estimates for the costs of inaction. Even if the lower-bound estimates are perhaps too conservative, with a bias towards health effects, they will in many situations encourage more rather than less abatement effort. Reducing emission loads will also tend to bring relief for the intangible assets of biodiversity and nature.

Making the best use of environmental science and modelling helps to make environmental protection and precaution a priority. Producing cost estimates should not be left to economists alone, but should rather be seen as a starting point for a broader discussion, featuring also the relevant expertise in health, ecology, demography, modelling and science. Well researched estimates, based on interdisciplinary collaboration, can strengthen some of those scattered and diffuse interests, which during the ordinary processes of policy-making have difficulty making their voices heard.

23.1 Introduction

The first volume of *Late lessons from early warnings* reminded us that:

'The costs of preventive actions are usually tangible, clearly allocated and often short term, whereas the costs of failing to act are less tangible, less clearly distributed and usually longer term, posing particular problems of governance. Weighing up the overall pros and cons of action, or inaction, is therefore very difficult, involving ethical as well as economic considerations, as the case studies illustrate' (EEA, 2001:3-4).

In the decade since its publication, there has been considerable interest in addressing and understanding the possible costs of inaction. The Stern review on 'The Economics of Climate Change' for the UK government is a prominent example of how an economist specialising in risk assessment was able to provide credible estimates of the global costs of failure to prevent climate change (Stern, 2007). Stern warned that the likely consequences of continued, unabated global warming would be in the magnitude of 5 % of global GDP annually. He estimated that the costs could rise as high as 20 % when including non-market costs for health and environment and if we factor in the effects on developing countries in an equitable way. In contrast, according to Stern's review, the average expected costs of mitigation for stabilising greenhouse gas concentrations are likely to be about 1 % of GDP annually, while unlikely to exceed 3.5 % of GDP annually.

The Stern review has been subject to much attention and scrutiny, mainly over the so-called 'discount rate' (a measure of how to value costs and benefits in the future — see section below) it used. However, its basic finding that the costs of preventive action were less than the harm caused by inaction was broadly accepted. The Review influenced the political debate, and the UK government subsequently passed the Climate Change Act, which called for an 80 % reduction in greenhouse gases by 2050. The Stern review also influenced the European Council in its decision in March 2007 to embark on a more active climate policy.

The subject of the 'costs of inaction' has subsequently become a recurrent topic in deliberations over climate policy, and more recently, with respect to biodiversity. The OECD (2008) has provided an overview of studies addressing the costs of inaction for a range of other environmental issues, including air pollution, water pollution, natural resource depletion and industrial accidents. To understand the costs of inaction, one must express in monetary terms the damage that will be caused if no or limited intervention is agreed. Stern did this by taking a risk-analysis approach, which unlike the more static cost-benefit approach, emphasised a range of outcomes and the uncertainties involved in the calculations.

It is essential to understand that under an economics perspective the 'costs of inaction' are simply analogous to the benefits that can be obtained with proper controls (OECD, 2008:49). Addressing the 'costs of inaction' involves the same methods used to account for benefits in conventional cost-benefit analysis. They have in recent years been underpinned by the availability of computing and modelling capacities, that better allow environmental scientists and environmental economists to account for the complexities and uncertainties at stake, and hence to integrate a precautionary perspective.

This chapter reviews some of the methodologies used to produce estimates for the costs of inaction. Not all the case studies here will match chapters in *Late lessons from early warnings* Volumes 1 and 2. Instead, the case studies provide a generic perspective, which will be of some relevance to historical, current and future case studies.

The first case study we look at is the phasing out of lead in petrol. We use this example as a starting point for a discussion of the subtle differences between scientific proof and scientific evidence. In our second case study, we turn to the issue of nitrates in drinking water in order to show that costs of inaction can be estimated even when there are uncertainties over the long term effects of inaction. In the third case study, we explain some of the methodological controversies that have raged over how to account for the costs of inaction related to mortality risks, which is relevant for air pollution for example. By way of illustration we present in separate boxes monetary estimates of the environmental burdens related to lead and mercury, SO_2 with other air pollutants as well as to ozone-depleting substances (ODS).

23.2 Should we require proof or evidence to account for the costs of inaction: insights from lead phase-out

Heinzerling, Ackerman and Massey (2005) provide a concise overview of the role of scientific and economic analysis in eventually ending the use of

lead in gasoline in the United States. It was research by Herbert Needleman (see Chapter 3 on lead in petrol) on the relationship between lead levels in children's blood and cognitive impairment that made it possible to state in quantitative terms the loss of IQ involved. Heinzerling et al. describe how economic analysis eventually came to the rescue of the lead phase-out by translating this IQ loss into monetary figures. Lower IQ could be shown to result in lower life-time income, which could in turn be translated into an implied cost of inaction (see also Nichols, 1997).

This approach differed from the simple 'willingness-to-pay' studies that previously had been referred to by economists. In these willingness-to-pay studies, parents were asked how much they would spend to prevent exposure of their children. These studies were problematic in that they expected firm monetary preferences to emerge for a problem the dimensions of which few parents would understand. The shift in focus away from willingness-to-pay studies meant that the findings of the lead scientists were used directly in economic assessments of the costs of inaction. The human bones absorb lead, and Needleman's breakthrough was to measure the stocks of accumulated lead in the first 'baby' teeth of children, rather than the level of more recent lead exposure in their blood. One challenge was to establish to what extent blood lead levels reflected ambient concentrations of lead in air. The lead case is described in more detail in Box 23.1.

Today there are numerous hazards and challenges for which we have only limited scientific evidence, and for which it will take many years to accumulate the same understanding as was reached in the case of lead. Economists trying to estimate the costs of inaction will often ask whether there is 'scientific proof', but it must be acknowledged that there is usually not a simple answer to this question. Economic analysis should therefore consider the less certain levels of knowledge that make up scientific evidence in order to explore the magnitude and risks of potential impacts suggested through early warning signals.

We sometimes tend to think of knowledge and lack of knowledge in a simplistic way, believing that research is a process that can transform lack of knowledge into indisputable proof. More realistically, our knowledge base is less clear-cut. It often has a focused nucleus of complete understanding, with a surrounding area where many linkages and relationships are not understood with the same rigour and underpinning. Early warning signals are often located in this blurred area where linkages are not fully understood.

The Treaty on the Functioning of the European Union (TFEU), which establishes the precautionary principle as one of the guiding principles of environmental policy, does not generally mandate a requirement for scientific *proof*. For instance, TFEU art. 114 no. 5 — the so-called environmental guarantee — allows a Member State to introduce unilaterally more stringent measures in areas where there is 'scientific *evidence*'. The precautionary perspective (TFEU art. 191) is really about acting *in the absence* of scientific proof (see Chapter 2 on the precautionary principle).

Although Needleman's research was able to document a relationship between low doses of lead and IQ impacts on children, not all aspects of lead exposure were fully understood. Today we can take advantage of computerised environmental models to account for blood lead concentrations as a result of the lead accumulation over time in the various compartments of the human body. In this way, we can clarify the link between low-dose exposure and the resulting, age-dependent blood lead in children (Pizzol et al., 2010). But the use of such tools remains the exception. For example, the chemical PCB is a substance for which the abatement costs are now counted in billions of euros (von Bahr and Janson, 2004) while welfare economic damage costs are largely unknown. Therefore, when it comes to substances for which the knowledge base is less developed, it will be necessary to explore the potential costs of inaction by relying on evidence from early warning signals. These harm costs could for instance be calculated by looking first at the already-proven impacts of high doses in the work environment, and then scaling linearly to low doses.

23.3 Using warning signals to estimate the costs of inaction: risks from nitrate in drinking water

The European Union's 1980 directive on drinking water quality (80/778/EEC) introduced a maximum admissible concentration value (MAC) for nitrate contents in drinking water, based directly on WHO guidelines. Drinking water with MAC levels of nitrates is suspected to increase the occurrence of *methaemoglobinaemia* in vulnerable individuals, particularly babies (leading to the potentially fatal phenomenon of 'blue' babies, cf. NSW, 2006). At MAC-concentration levels, drinking water nitrate will double the amount of nitrates that the average person would ordinarily consume from other sources.

Box 23.1 Costs of inaction on lead and mercury

Why does lead matter?
Lead is a potent and pervasive neurotoxicant that travels widely throughout the body once ingested or inhaled, and affects virtually every organ or system in the body (Meyer et al., 2008). Children and pregnant women have a higher absorption rate of lead due to constant bone remodelling, which arises from skeletal development (Barbosa et al., 2005). The impacts upon children are profound, because not only is lead more easily absorbed, it is also more damaging to the developing nervous system. Even extremely low Bloodstream Lead Levels (BLL) can have significant impacts on future academic achievement (Miranda et al., 2011), a fact that was a material consideration in the 2005 US Centre for Disease Control's recommendation that there is no safe BLL for children (ACCLPP, 2012). These changes in development are irreversible. Exposure to even low levels of lead during pregnancy will cause permanent and irreversible developmental harm, which manifests itself in lower educational achievement in later stages (Lourdes et al., 2006).

The damages arising from lead have been described as nothing less than a 'catastrophe' for public health (Landrigan, 2002). Lead damages almost every biochemical process in the human body (Gidlow, 2004; Needleman, 2004). It damages fertility and neuropsychology, as well as distorting enzymes, structural proteins and mitochondrial cristae. It also pushes out calcium from natural neuron signalling processes (see Box 3.1 on children and lead in Chapter 3).

Recognition of the problem
Lead has been recognised as a health problem since Roman times (Reddy and Braun, 2010). But it was not until 100 years ago that an Australian doctor identified the critical vulnerability of children to lead (Taylor et al., 2010). But despite multiple early warnings, lead continued to be used as an additive in paint for many years in Europe, America and Australia.

The most harm was caused by the use of lead as an additive to motor vehicle fuel, which resulted in widespread environmental exposure and serious health impacts for hundreds of millions of people. Lead was first added to motor fuel in the 1920s, and it took many decades before developed nations began to phase out the practice.

The history of the use of lead as an additive in petrol is a sad testament to the desire of companies to make profits regardless of the consequence to human health. Other, much less harmful alternatives to lead were available to the petrochemical industry, but they were not adopted because they were not patent-protected, and would therefore have greatly reduced profits for the sector (Ackerman et al, 2005). The global harm caused by leaded gasoline is undoubtedly high, not least because extremely high BLL have been measured in different cities. For example, the average BLL of the sampled population in Bangkok in the 1980s and 1990s was 40ug/dl, levels at which effects such as headaches, slowing of motor nerve conduction and anaemia may occur. Death may occur at BLL of about 100ug/dl (Olson, 2004).

In 1981, the EU introduced a limit of 0.4 gram of lead per litre of leaded gasoline. Use of lead became less attractive following the requirements for installation of catalytic converters in cars. Germany introduced different tax levels for leaded and unleaded petrol in 1985, an approach copied in several other Member States (Hammar and Löfgren, 2004). This meant that by the time the European Union prohibited use of leaded gasoline in 2000, there was little protest from any quarter. As of January 2011, only six countries continue to allow leaded gasoline: Iraq, Yemen, Algeria, North Korea, Burma and Afghanistan.

Economic impacts
In the US, economic analysis was instrumental in convincing the Reagan administration of the desirability to phase out lead in petrol (US EPA, 1985). These analyses explored the loss of IQ in children exposed to lead, a phenomenon that had been demonstrated by scientists. Cognitive impairment of children could be shown to result in reduced expectations for life-time income (Salkever, 1995). Statistically, a loss of one IQ-point can be shown to cause a loss in life-time income of 2-3 % (highest for females). There is no lower threshold for the lead-induced cognitive impairment of young children, so all emissions impair cognitive function (Schwartz, 1994).

Because the harm from lead relates to a future stream of income over a lifetime, the discount rate used to convert future earnings into a net present value decisively influences the final monetary estimate for the

Box 23.1 Costs of inaction on lead and mercury (cont.)

harm caused. In studying the harm cause by lead, US economists used a discount rate of 10 %, while an EU study abstained from any discounting ([1]). Early childhood IQ-losses, such as those that arise from undernourishment or poisoning, can be significant. IQ levels have been demonstrated to influence dramatically the relative affluence of nations, and are therefore a social good (see Figure 23.1).

When using a *social* discount rate of 1.4 % (cf. Stern, 2007; European Commission, 2008) the result is a damage estimate of EUR 1.50 per gram of lead emitted in urban areas (Andersen, 2010). The discount rate is not the only factor that influences the cost estimate. Estimates of the cost of damage can also be greatly influenced by assumptions about how lead enters the food chain ([2]) and by assumptions about so-called 'resuspension'. Resuspension is the process by which lead emitted years ago is deposited on the soil, but then blown up into the air again by the wind.

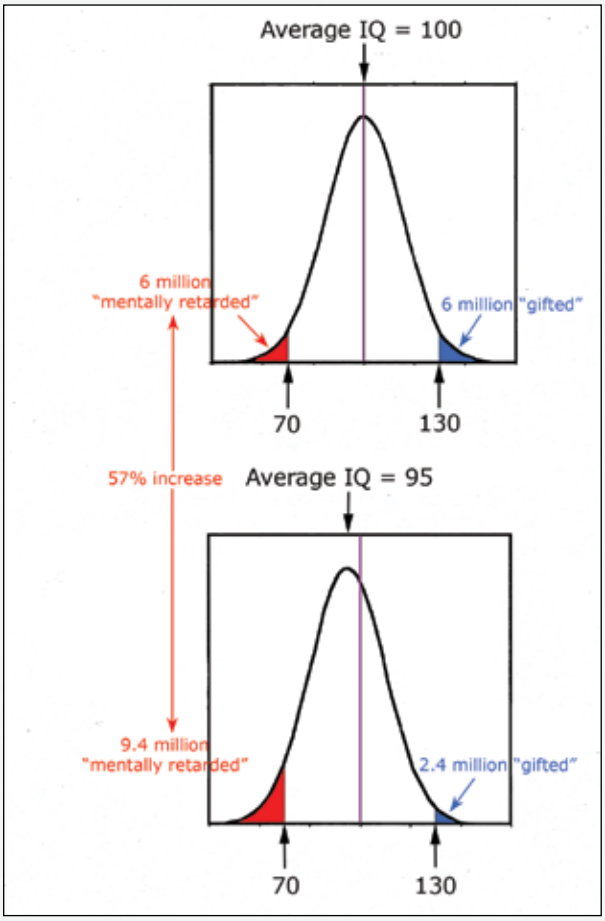
Considering that until the 1970's, gasoline contained about 1 gram of lead per litre, valuing lead-associated IQ damages at EUR 1.50 per gram (EUR 1.50 per litre) implies significant annual costs of about 4–6 % of GDP in EU Member States.

Today, lead emissions in the European Union have decreased greatly, and the magnitude of damages from new emissions, mainly from waste incineration and industry, amounts to less than 0.1 % of GDP. Still, further reduction of lead in products (e.g. paints, toys) remains desirable to prevent continued harm to children.

All the lead emitted during the 20th century is dispersed across the environment, contributing each year to renewed harm as some of the lead finds its way back into our atmosphere. Many urban areas have chronically high levels of lead in top-soils. This is sufficient reason to recommend the resurfacing of playgrounds and areas of high activity for young children, as the intake of even a few micrograms may suffice to induce IQ-damage.

Economic damage estimates are based on scientific research on the impact of human exposure to a particular substance. But not all risks are factored into cost-benefit analysis. For example, the ability of lead to trigger cardiovascular diseases in adults (hypertension, nonfatal heart attacks and premature deaths, cf. Menke et al. 2006) has not been included in the above-mentioned damage estimate. Thus, some damage estimates are lower-bound conservative values (Gould, 2009). This tendency for cost-benefit analysis to lean toward conservative values further strengthens the case for adopting a precautionary perspective.

Figure 23.1 Diminished IQ stunts economic development



Note: This graph shows the social impact of a drop in average population IQ. For example a 5-point drop could raise by 57 % the number of individuals considered neurologically impaired, with implications for social and economic development. It also significantly lowers the number of intellectually gifted children.

Source: www.ourstolenfuture.org.

([1]) The US EPA study due to its use of a private discount rate implied a damage cost of 8 US cents$_{1983}$ per gram lead, while the European study with no discounting implied a damage cost of 5.9 EUR$_{2004}$ per gram lead.
([2]) Spadaro and Rabl (2004) indicate a ratio of 1:25 between inhalation and ingestion (intake) from food.

Box 23.1 Costs of inaction on lead and mercury (cont.)

Why does Mercury matter?
Mercury has a long atmospheric lifetime (6–18 months), which means that once it is released from a source anywhere in the world, it can be transported globally, hence its characterisation as a 'global pollutant'. After deposition in ground or water, mercury can be transformed. The process by which this happens is primarily microbial action, which turns mercury into methylmercury. Methylmercury can 'accumulate' as it progresses up the food chain. For example, an animal could eat a smaller animal that has eaten mercury, exposing the larger animal to all the mercury consumed by the smaller animal in its lifetime. This results in ever higher levels of mercury being found in larger predator animals. Human health can be adversely affected by this process of accumulation if organisms with high concentrations of mercury are ingested. This is a particular problem in consumption of predatory fish, notably tuna and shark (Shimshack and Ward, 2010). Pregnant women are at particular risk, as mercury readily passes through the placenta, concentrates in umbilical tissues and leaches into breast milk. Methylmercury is a developmental neurotoxicant , and high environmental exposure to this compound is associated with a statistically significant reduction in IQ in developing children.

Recognition of the problem
Despite the official acknowledgement of Minamata Disease (a sickness affecting people exposed to methylmercury in the Japanese city of Minamata, see Chapter 5) in the spring of 1956, success in tracing the cause to the mercury discharges from an industrial facility took several years. It was not until 1968 that the Japanese Government 'announced its opinion', that factory effluent was directly responsible. An official apology was finally made at the 50th anniversary of the discovery of the disease (Japan Times, 2006) ([3]). Although large-scale acute cases of mercury poisoning are relatively rare, the more general result of lower IQ due to the developmental neurotoxicant effects of methylmercury continues.

Economic impacts
Mercury damages the developing brain and reduces IQ, just like lead (see above). And like lead, it is a substance that has only harmful effects. This is in contrast to other metals that are also toxic at high doses, but of which the human body needs a certain minimum to survive. Several studies have attempted to calculate the costs for diffuse poisoning (i.e. poisoning from many different sources) by mercury. Despite the wide uncertainty in these figures, they demonstrate that the impact is far from trivial.

Axelrad et al. (2007) created a 'dose-response' model to assess the effect on IQ of each additional 'dose' of mercury exposure by performing an integrative analysis of studies from New Zealand, the Seychelles and the Faroe islands. Their central estimate is that for every 'part per million' increase in mercury in the hair of an expectant mother, the child suffered a 0.18 point decline in their IQ ([4]). Concentrations of mercury in hair can be converted into blood concentrations, or concentrations in the umbilical cord, using established factors. As is the case with other mercury studies, the authors assume that there is a minimum threshold (0.1 ug per day per kilogram bodyweight) below which no effects of mercury occur.

For low doses, the time window during which the brain is affected by mercury needs to be considered. The sensitivity of the brain to mercury is greatest during the early development of the body. The epidemiological studies all assume that once a person is exposed to mercury, the effects on their IQ are both measurable and irreversible, remaining with the person throughout their lives. Since the dose-response function refers to maternal hair concentration and effect on children's IQ, it implicitly includes the effect of mothers' diet during pregnancy and early infancy of her child while she is nursing.

Spadaro and Rabl (2008) have calculated the marginal impact of low-dose mercury emissions. Applying an estimate for the monetary value of a 'global' IQ-point (i.e. the value that an extra IQ point brings a person in terms of lifetime income), Spadaro and Rabl report a marginal damage cost per kilogram of mercury emitted of about USD 1 500. If not assuming any threshold to the impact of low doses, the cost is reported to be USD 3 400 per kg. These estimates are conservative compared with the average damage cost of over USD 6 000 USD per kg implied by one US EPA study (where a 33 tonne change in mercury emissions would cost USD 210 million), which used a different methodology and did not have any threshold for pre-natal exposures (Griffiths et al., 2007).

A study in Greenland, where three quarters of newborns have elevated blood concentrations of mercury, even though there are hardly any local emissions, estimated a damage of 59 million dollars per year (Hylander and Goodwin, 2006). This translates to about USD 59 000 for each new born child. The traditional diet in the Arctic region of fish and sea mammals serves as a sink for global mercury emissions by means of the accumulation process described above. Global mercury emissions are projected to increase by 20 % by 2020 (relative to 2005 cf. AMAP, 2011:143).

([3]) 'Koizumi issues official Minamata apology', The Japan Times Online, 29 April, 2006, http://search.japantimes.co.jp/cgi-bin/nn20060429a4.html.
([4]) A part per million is roughly equivalent to the concentration of a single drop of a substance in 50 litres of water.

In addition to a MAC-value of 50 mg NO_3 per litre, the directive introduced a complementary 'guide value' of 25 mg NO_3 per litre. When revising the directive in 1998, the precautionary guide value for nitrate in potable water was deleted and only the MAC-value retained. The main reason for abandoning a guide value was the absence of scientific proof to underpin it.

Starting with the introduction of the drinking water directive (80/778/EEC), the European Union introduced several initiatives to reduce environmental pressures from nitrogen. One such initiative is the nitrogen application ceiling for agricultural land, and the associated regulations on animal manure that were introduced with the Nitrate Directive (91/676/EEC). The Urban Waste Water Directive (91/271/EEC) also introduced constraints for emissions of nitrogen from sewage and industrial effluents. The protection of drinking water was not the only goal in these efforts to reduce nitrate leaching and emissions. Policy makers were also motivated by concerns over eutrophication in shallow and open coastal waters. Over the years, many conflicts have appeared over the costs associated with both these directives. Farming interests in particular have requested derogations and exemptions from regulatory requirements aiming to reduce nitrate loads.

No analysis has so far been carried out in Europe to explore how the possible benefits of these nitrate regulations compare to the costs of implementing them. The evidence that nitrates have adverse health effects is contested. In particular, there has been controversy over the extent to which gastric cancer is related to nitrate intake. The World Health Organization experts maintain that 'a link between cancer risk and endogenous nitrosation as a result of high intake of nitrate and/or nitrite and nitrosatable compounds is possible' (WHO, 2007:12). WHO normally requires at least three different studies establishing comparable evidence before accepting a link to exposure as conclusive, but in the case of nitrate there are few well-designed epidemiological studies in the international scientific literature.

Chronic impacts resulting from exposure over longer periods of time due to elevated levels of nitrate in drinking water have caused particular concern. In order to detect such influences, epidemiological cohort studies, which monitor health effects in a large population sample over a series of years, are required. Such cohort studies are both laborious and costly, and they often face difficulties with establishing the specific historical exposures. At present, the international scientific literature reports results only from two cohort studies regarding nitrate, and their results are ambiguous.

One study by Weyer et al. (2001) reports an increased incidence of bladder cancers in a population cohort of 10 000 women aged 55 or more from Iowa, US — a state with intensive agricultural practices, and high levels of nitrate in public utility water supply. A more recent study could not detect an increased incidence of bladder cancers in a population cohort in the Netherlands, presumably because nitrate levels in Dutch public water supply are rather low.

The Iowa study was published after the EU Council of Ministers abandoned the 25 mg NO_3 per litre guide value for nitrate in drinking water. Along with other types of studies, the Iowa study suggests that health effects can be detected well below the MAC-value and with lower thresholds of 15–25 mg NO_3 per litre. Hence there appear to be costs of inaction at stake, but a relevant question is: how significant are they?

The studies provide evidence from which nitrate health effects can be quantified. A quantification of these health effects is useful for estimating the *potential* costs of inaction — the risks in other words. For instance, the World Bank (2007) has used those studies that show a connection between ill health and nitrate exposure as a basis for providing rough estimates of the health benefits associated with reducing nitrate exposures. More detailed studies have taken a similar point of departure. Van Grinsven et al. (2010) explore the health risk costs related to colon cancer from nitrates, and arrive at a level of EUR 0.7 per kg of fertiliser nitrogen. Andersen et al. (2011), in the EU FP7 EXIOPOL project, explored different river catchment areas and arrived at site-specific estimates based mainly on figures for bladder cancers derived from Weyer et al. (2001). Their mean estimate is EUR 0.3 per kgN-loss, but they report higher health risk costs for specific Member States, for example the United Kingdom and Belgium at a level of EUR 1.3 per kgN. In some urban areas, the health risk costs are much higher. Consequently, it can be estimated that the risk costs of inaction on nitrate in drinking water amount to EUR 2.6 billion annually for the United Kingdom alone. This figure is comparable to a previous estimate for all external costs of UK agriculture combined (Pretty et al., 2001).

Since there are also other costs related to nitrogen, arising from the pollution of surface waters, ammonia evaporation, and greenhouse gas emissions of N_2O, these figures taken together

would suggest that a precautionary approach is warranted. The abandoned guide value for nitrate in drinking water also deserves reconsideration.

23.4 Air pollution: how to account for the mortality risks from inaction?

The willingness of individuals to make economic sacrifices in order to reduce potential statistical mortality risk is determined by their risk aversion. People make decisions to this effect with many everyday choices, such as adding airbags and other safety devices to their car. In the United States, the authorities have determined risk aversion by resorting to wage-risk analysis. This involves exploring what wage premiums individuals require for more risk-prone occupations, and on that basis have estimated the value of preventing a fatality at about 5.5 million USD_{1999} (1999 prices).

In Europe, there has been more emphasis on applying specific values comparable to figures used in transport economics for avoiding fatalities. Still, air pollution has a risk profile different from road traffic: The average road victim is middle-aged, while victims of air pollution are believed to be mainly people over 65 (Pope, 2002). While a road victim on average loses 30–40 years of life expectancy, the average air-pollution victim may stand to lose only a few years. The European Commission, on the basis of expert advice, has opted to value a statistical life (VSL) at 1.4 million EUR_{2000}, but has adjusted it downwards to about EUR 1 million for air pollution to account for the advanced age of the typical air pollution victim. Similar adjustments proposed in the United States were however met with public outcry and were branded as a 'senior death discount' (NYT, 2011).

In the economics literature, VSL remains the conventional metric for the valuation of statistical fatalities. In recent years, it has been challenged by the adoption of more conservative estimates, based on the specific number of life-years lost. There has therefore been a debate over whether it would be reasonable to exchange VSL-figures with VOLY: the value of a life-year (Hofstetter and Hammitt, 2002). Questions that have been addressed include whether life-years towards the end of an individual's life should be valued more highly than an average life-year (as more precious) or rather should be valued lower, as reduced health and vigour is likely to reduce life-quality. VOLY-values can be derived schematically from VSL, by assuming that the average VSL represents a loss of life expectancy of 3–40 years as in road traffic.

OECD (2006) guidelines for environmental cost-benefit analysis recommend the use of VSL for acute mortality (as in road transport) and to introduce VOLY for cases of so-called 'chronic' mortality, (the result of elevated exposures to harmful substances over longer periods of time), such as air pollution or nitrate in drinking water (See Box 23.2 on SO_2 and other air pollutants for results obtained with the OECD approach).

In contrast, the Science Advisory Board of the US EPA maintains that the only solid value available for quantifying risk aversion in monetary terms is the VSL (US EPA, 2010:12). The obvious implication of the US approach is that it accords a higher monetary value to environmental risk reduction than Europe. Unfortunately, US economists are facing decreased political acceptance of the recommended approach. This was reflected in the recent decision by the Obama administration — acting under political pressures — to withdraw proposed restrictions on ozone pollution, despite the high costs of inaction.

VOLY-based figures provide perhaps only a lower-bound estimate for the benefits of action, but in many cases these conservative values are already sufficient to justify abatement action. The European Commission's impact assessment of the thematic strategy on air pollution (CAFÉ; Clean Air For Europe cf. AEA, 2005) programme decided to tackle the ambiguity over VSL and VOLY values simply by reporting different sets of benefit estimates. It then left it to policymakers to make a decision based on these two different pieces of information. While the most conservative estimate suggests that air pollution costs account for roughly 3 % of GDP in EU-25, the highest estimate amounts to 5 % (see Box 23.2 on SO_2 and other air pollutants). The acknowledgement that economics may not offer a mature consensus corroborates the role and significance of the precautionary principle. The Stern review of climate change reflected similar uncertainties, reporting different estimates, rather than presenting one specific economic figure.

23.5 Should we accept a discount on costs of inaction arising in the future?

The costs of inaction will often only be felt in the future. This is certainly the case for the impact of lead on IQ-loss or the health implications of living with high levels of air pollution and drinking water nitrate. In the case of the skin cancers induced by the ozone 'hole', impacts are expected to peak only after

Box 23.2 Costs of inaction on sulphur dioxide (SO_2) and other air pollutants

Why does air pollution matter?
Sulphur dioxide (SO_2) is a colourless, pungent gas that is a by-product of combustion at power plants, and also arises from natural sources such as volcanic eruptions. SO_2 emissions are a particularly important issue because they are produced in the combustion of coal, which is a major component of the electricity generation systems of many major economies (see Figure 23.2).

Sulphur dioxide has many potential environmental impacts, including its conversion to various acidic compounds (sulphuric acid, sulphurous acid) that can damage tree, plant and animal life (see EEA, 2001, Ch. 10 on sulphur dioxide, lungs and lakes). The converted and hence secondary formation of particulates (for example SO_4) after transport and chemical action cause severe health problems too.

Health impacts
Inhaled sulphur dioxide readily reacts with the moisture of mucous membranes to form sulphurous acid (H_2SO_3), which is a severe irritant. People with asthma can experience increased airway resistance with sulphur dioxide concentrations of less than 0.1 ppm when exercising. Healthy adults experience increased airway resistance at 5 ppm, sneeze and cough at 10 ppm, and experience 'bronchospasm' at 20 ppm (ATSDR, 2011).

The further risk posed by SO_2 is in its subsequent conversion to sulphate particles, and as a precursor to PM (particulate matter), contributing to ill-effects caused by PM_{10} (particulates of larger size) and $PM_{2.5}$ (particulates of smaller size). A meta-study of the mortality effects of ambient particulate sulphates demonstrates a strong correlation between mortality and the atmospheric density of the pollutant (i.e. the amount of pollutant present in the air, Smith et al., 2009).

Recognition of the problem
The first deadly impacts of an air pollution episode had been reported as early as 1929 in Wallonia, Belgium. Another significant and widely-recognised case of health impacts from sulphur dioxide was the Great Smog which affected London in 1952, causing an excess mortality of as many as 13 000 deaths (Bell et al., 2003). The smog was caused by a high level of air-borne pollutants, which remained in the city due to unusually still weather conditions. The response to this event was to propose remedies such as increasing the height of chimneys. Whilst this alleviated the problem, it also dispersed the pollution more regionally, and caused the acidification of rain, lakes and rivers in Scandinavia.

Japan also suffered badly from sulphur dioxide pollution in the three decades of industrialisation that started in about 1950. In all, more than 100 000 people were registered by the Japanese ministry of the environment as having suffered health impacts as a result. Sulphur dioxide pollution, known at the time as Yokkaichi Asthma, is listed as one of the four 'big pollution diseases of Japan' (Committee, 1997).

Economic impacts
In the 1970s, some claimed that the problem of sulphur dioxide pollution were not severe enough to warrant expensive action. Some even called it a 'million dollar problem with a billion dollar solution' (Opinion, 1977). The link between atmospheric concentrations of sulphur dioxide and sulphate to increased mortality rates was only gradually accepted. However, when one factors in not only the costs of morbidity (the sickness caused by sulphur dioxide) but also those of mortality (the deaths caused by sulphur dioxide) it is clear that the health costs of inaction on air pollution damages should indeed have been counted in billions too.

Pope et al. (1995, 2002) showed how increased rates of mortality are consistently associated with high ambient levels of pollutants. Most of the deaths were seen to occur in response to chronic exposures over extended time periods, rather than in response to shorter periods of exposure. Their findings were based on a population cohort of more than 500 000 individuals, who had been interviewed about health status, smoking habits and other potentially confounding variables. The American Cancer Society had obtained individual death certificates from deceased participants in the cohort, which allowed control for death caused by air pollution exposure. The US EPA decided to require a complete reanalysis of the data by an independent research team before finally accepting the findings — it led to the same results.

In 2005, the European Commission commissioned a comprehensive cost-benefit analysis when preparing its Thematic Strategy for Air Pollution. It also invited the World Health Organization (WHO) to review the

Box 23.2 Costs of inaction on sulphur dioxide (SO_2) and other air pollutants (cont.)

health effects evidence available. Because of the methodological debate over the appropriate valuation metric for the risks to human lives (see discussion in main text of this chapter), the assessment presented different estimates. The lowest estimate, based on a VOLY metric, came to a total cost of air pollution for the EU-25 of EUR 276 billion, equivalent to 3 % of GDP. The highest estimate, based on a VSL metric, came to a total cost of EUR 427 billion, close to 5 % of GDP (AEA, 2005).

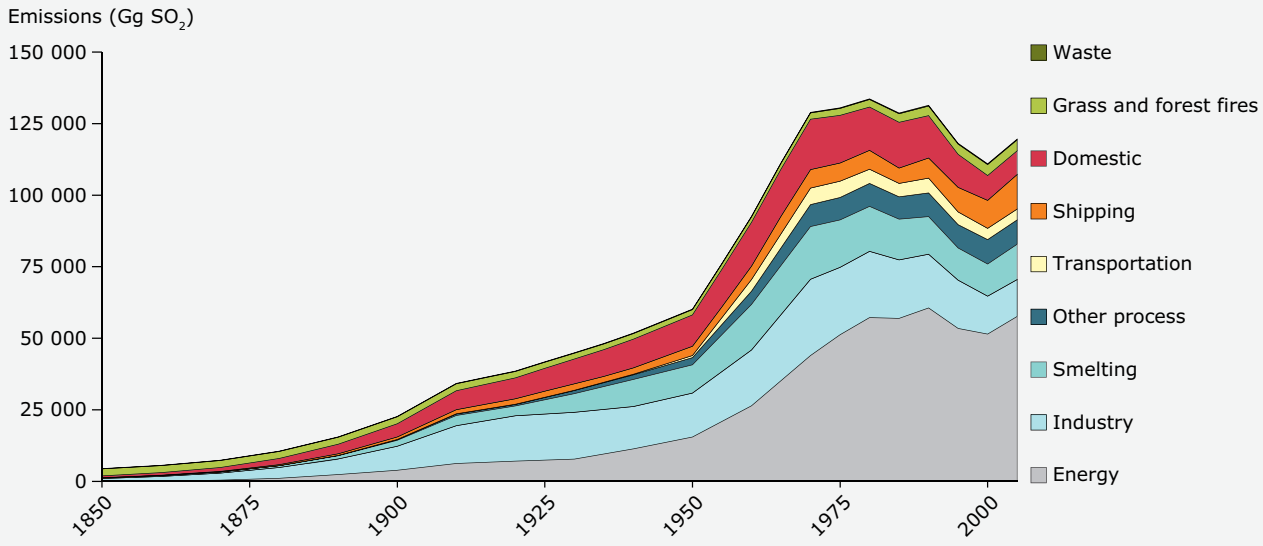
The costs were seen to differ considerably among Member States. In Poland and other new Member States, the air pollution costs (using 2000 as the reference year) varied between approximately 15 % and 22 % of GDP depending on the metric (AEA, 2005). These costs referred to the combined exposure to particles from SO_2, NO_X and primary PM.

Although these figures do not include all relevant costs of inaction — for instance in relation to biodiversity — the figures suffice to justify further action to reduce air pollutants. While the damage cost per kilo of SO_2 ranges from EUR 5 to 9 per kg SO_2, marginal abatement costs start from below EUR 1 per kg SO_2 (Rive, 2010). Theoretically, it would be desirable to pick the lowest-hanging fruit first and focus on the most cost-effective measures, but in reality there are large benefits for nearly all efforts regarding SO_2 abatement.

A cost-benefit assessment becomes highly complex when multiple types of emissions are considered jointly (SO_2, NO_X, VOC, O_3, NH_3 and PM). There are non-linearities at play in the atmospheric transport and chemistry, for instance relating to ozone, which must be taken into account. Some integrated assessment models have been created to study these situations that feature complex mixtures of emissions. They typically produce scenarios for different levels of pollutants (e.g. SO_X, NO_X) and come up with a best-cost solution for meeting various environmental outcomes. The technically feasible reduction of air pollution has been estimated at EUR 56–181 billion equivalent to 0.6–2 % of GDP in EU, which are the costs that could be avoided by introducing appropriate controls (AEA, 2005).

Japanese industry was a pioneer in air pollution abatement. In 1975, investment in air pollution abatement accounted for 18 % of Japanese capital investment and 6.5 % of GDP. The OECD (1977) has reviewed the experience and concludes that 'the impact of relatively high pollution abatement costs on macro-economic magnitudes, such as GNP, employment, prices and foreign trade is practically negligible'. In fact, these investments accelerated technological innovation, raised product quality and lowered technical costs. Even today, Japanese companies control many of the patents and licenses for air pollution control equipment, and benefit from sales globally, demonstrating the economic significance of being a pioneer in environmental technology.

Figure 23.2 Global SO_2 emissions 1850–2005 by end-use sector



Source: Smith et al., 2011.

60 years. These delayed effects present a particular problem for estimating the costs of inaction.

To allow for the comparison of monetary estimates at different points of time, economists typically use 'discounting' techniques, whereby all estimates of future costs are discounted — or reduced in absolute terms — into net present values. There are two textbook reasons why economists assume the future is not worth the same as the present to an investor. Firstly the investor may not be alive when the return on the 'investment' is made, and secondly, the prospect of continued economic growth and technological progress means an investor expects to be richer when the return is made, meaning that the return will have less value in the future. For this reason economists apply a discount rate, reflecting mainly these two aspects, and adjusting them for time preferences and consumption value.

To many non-economists, the implied shrinking of future values with the discounting technique is at odds with the core idea of intergenerational equity that is central to sustainability. A related problem is that many economists are applying discount rates that are typically used in the corporate sector — rising up to as much as 10 % — without reflecting on the specific context of environmental challenges.

As pointed out in the Stern report, time preference discounting is less relevant for a society than to an individual. Firstly, society is not mortal. There is only a small risk that societies would be discontinued, whereas an individual investor faces a much greater risk. For this reason, Stern recommends representing this risk of social 'mortality' with a tiny discount rate of 0.1 %. As for the second aspect of discounting, the consumption discount rate that seeks to compensate for continued economic growth in the future, Stern maintains it should be based on expectations for future economic growth, net of inflation. Only a very small number of countries have expectations for annual economic growth rates of 10 %. In Stern's analysis of climate change policy with the PAGE model, he simulated many different trajectories of economic growth for the future decades and came to an average expectation of 1.3 % per annum. Stern's review argued that even when discounting the stream of future benefits, the aggregate sum of avoided damages — from 5 to 20 % of annual consumption — would well exceed the involved costs. Table 23.1 illustrates how the social cost of carbon depends crucially on the discount rate chosen.

Many environmental projects have much shorter time horizons than climate change policy. For example, investments in sulphur scrubbers for air pollution abatement have only a 10 or 20 year lifetime. With shorter project lifetimes than in climate change policy, there will be less 'shrinking' of the future, even for the lifetime-loss of income of the lead-poisoned child (see estimates in Box 23.1 on lead). For mortality risks, discounting is not very important because the VSL must be adjusted upwards for expected economic growth, which will cancel out the consumption component of the discounting, leaving only the 0.1 %.

Table 23.1 Effect of the discount rate on the estimated costs of inaction

Discount rate (%)	Discounted costs of inaction (% of GDP-equivalents)
1.3	14.7
1.8	10.6
2.3	6.7
2.8	4.2

Source: Stern, cited from OECD, 2008:96.

23.6 Concluding remarks

We have discussed above some of the problems with the willingness to pay model of estimating the costs of inaction. In what has become a classic discussion of this model, Diamond and Hausman (1994) raise two other difficulties with willingness-to-pay and the so-called 'contingent valuation method' (CVM) it uses. Firstly, because stated willingness-to-pay is hypothetical, there is an inherent risk that any respondent will overstate their preferences, neglecting their level of income and therefore their real ability to actually pay. Secondly, there is the tendency of willingness-to-pay results to be inconsistent across different surveys. For example, the willingness-to-pay for cleaning up one lake might be similar to the willingness to pay for cleaning up five. People may simply express a desire to contribute EUR 50 for a good purpose, but have no specific individual preference as to the amount of the public good in question. These are strong methodological critiques to the use of monetary estimates derived from CVM.

Europe has acted cautiously in mandating formal requirements for cost-benefit analysis. At Member State level, there is a fairly limited tradition as part of formal legislative processes. Although the concept has found its way into common-day language across the European continent (Germans for instance speak of 'Kosten-Nutzen analyse'), it is perceived by many as a somewhat American-style approach to policymaking processes. The Dutch felt compelled to rename it as

Box 23.3 Benefits of early action on ozone depleting substances

Why do ODS matter?

The ozone layer absorbs most of the high-frequency ultraviolet radiation that could cause damage to life on Earth. Ozone Depleting Substances (ODS) are chemicals that can survive long enough in the atmosphere to migrate to ozone-rich areas, in the upper atmosphere, some 25 km high. At this altitude, ODS then undergo reactions that break down ozone molecules (see also Figure 23.3). If the emission of ODS is allowed to continue or increase, the ozone layer will reduce in thickness, increasing the quantity of harmful UV radiation that reaches the surface of the Earth (Molina and Rowland, 1974 — see also Chapter 7 on halocarbons and the ozone layer in *Late lessons from early warning* Volume 1 (EEA, 2001)).

Exposure to UV radiation has significant health implications for many forms of life, including humans. Excessive exposure is linked to increasing rates of skin cancer, cataract development and reduced capacity to resist bacterial and viral infection (WHO, 1994). Insufficient exposure can also create problems, notably by leading to vitamin D deficiency, and damaging crop productivity.

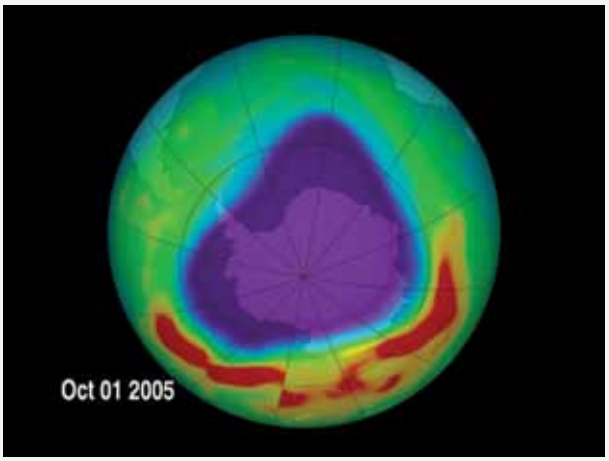
UVB radiation is a major causal factor in the development of melanoma and non-melanoma skin cancer (NMSC). A reduction in the thickness of the stratospheric concentration of ozone allows a greater proportion of UVB radiation to reach the Earth's surface, and will generate an increase in UVB-related cancers. Although NMSC is not as serious as melanoma, it accounts for the majority of skin cancers.

Recognition of the problem

Scientist Richard Scorer of Britain's Imperial College was a respected environmentalist, but sided with industry on the safety of CFC's, claiming in 1975 that 'The only thing that has been accumulated so far is a number of theories' (Roan, 1989:61).

In spite of scepticism about the ozone depletion theory from some quarters, the case for limiting the production and emission of ODS is often viewed as a *cause célèbre* for international agreements. This is because it was one of the first cases of successful coordinated, international action on phasing out a chemical (CFCs were the first of the ODS to be banned) that was in widespread use, and that was shown to be causing an environmental impact on a global scale. The Montreal Protocol entered into force on 1 January 1989, and was recognised by Kofi Annan in 2003 as 'perhaps the single most successful international environmental agreement to date'. It went on to become the first international treaty to be universally ratified, on 16 September 2009, by 196 countries, and has shown demonstrable success in achieving its stated objectives. This means that the ozone layer should return to its pre-1980 levels sometime between 2050 and 2075 (UNEP, 2009a).

Figure 23.3 Region of exceptionally depleted ozone in the stratosphere over the Antarctic



Source: NASA/Goddard Space Flight Center Scientific Visualization Studio.

A study undertaken in 2009 attempted to predict the future that was avoided by the Montreal Protocol and by subsequent international agreements on ozone-depleting chemicals (Newman et al., 2009). The benefits of early action are starkly illustrated in the predicted 'World Avoided' ultraviolet (UV) radiation index compiled by that study (see Figure 23.4). This figure shows the impact of the increasing amount of harmful UV radiation that would have been permitted to reach the Earth's surface. Without international agreements to eliminate ODS production and emissions, the ozone layer could have reduced in thickness by as much as 67 % by 2065, with highly damaging consequences for humans and many other organisms.

Box 23.3 Benefits of early action on ozone depleting substances (ODS) (cont.)

Economic impacts

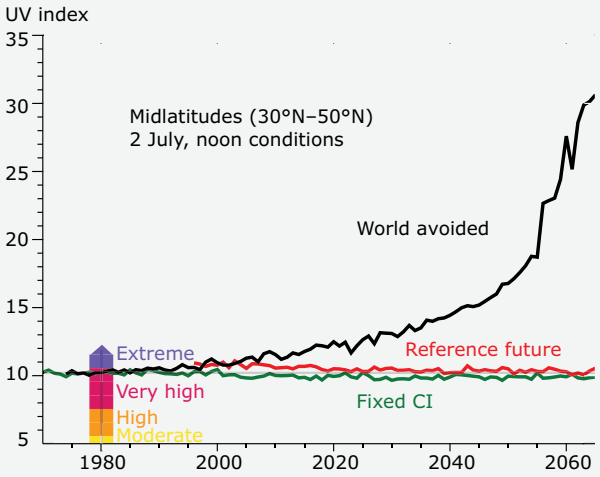
Health

Incidence of skin cancers in response to increased UV radiation are expected to peak about 60 years from exposure. Assuming prevention of a 48 % decrease in the ozone layer by 2050, UNEP (2009b) has estimated that more than 20 million skin cancers and 130 million cataract cases have been prevented globally as a result of the Montreal protocol.

The stabilisation of the ozone layer means that by 2050 annually about 47 000 skin cancer cases will be avoided in north-western Europe, although 14 000 additional cases of skin cancer are still to be expected as a result of the damage to the ozone layer that has already been done and the 60 year latency period (Slaper et al., 1998:83; Velders et al., 2001:8). 99.5 % of skin cancers are likely to be non-melanoma with a mortality rate of 1 %, while 0.5 % will be melanomas with a mortality rate of 24 %. The implication is about 500 fatalities avoided annually in north-western Europe towards the middle of the 21st century.

Assuming the same ratio between fatality reduction and other avoided UV-radiation effects (mainly health-related) as in US studies (Sunstein, 2007), these figures imply that annual benefits of early action from the Montreal Protocol are not less than EUR 3 billion for Europe. Scaling results for north-western Europe to all of the European Union must take into account higher exposure risks in southern Europe, with likely annual benefits of EUR 10–11 billion for EU-27.

Figure 23.4 The 'world avoided' UV index as a result of international agreement



Source: Newman et al., 2009.

Climate change benefits of early action

ODSs have had a considerable impact upon so-called 'radiative forcing', and therefore on global warming (Radiative forcing is a measure of the influence a factor has in altering the balance of incoming and outgoing energy in the earth-atmosphere system). It is estimated that 13 % of the present total global warming effect is due to the release from year 1750 to 2000 of Halocarbons and ODSs (IPCC, 2007).

The *avoided* impacts of ODSs on climate change are substantial. The combined effect in 1990 was 7.5 ± 0.4 $GtCO_2$-equivalent/year, or about 33 % of the annual CO_2 emissions due to global fossil fuel combustion (IEA, 2009). If we assume — in the absence of a Montreal protocol — a 3 % annual increase in ODS and halocarbon production, the 2010 emissions would have amounted to ~ 14 $GtCO_2$-equivalent/year. In other words the Montreal protocol has over two decades saved the atmosphere for about 215 $GtCO_2$-equivalent in emissions. Assuming that ODS had not been regulated separately, but also were to be counted under the Kyoto Protocol, these emissions, that stem mainly from developed countries, should have been offset and could have represented a cost of about EUR 2 150 billion (assuming a price of EUR 10 per tCO_2-equivalent) — about 0.5 % of annual GDP of OECD countries over these two decades.

The CO_2-equivalent of ODS-reduction is bigger than cuts required by developed countries under the Kyoto Protocol. Important is also the time delay achieved in relation to climate change. It will take between 7 and 12 years for CO_2-emissions to increase by the amount of ODS abated with the Montreal Protocol. If considering the full reduction achieved since the 'early warning' scientists Molina and Rowland (1974) first called attention to the ozone layer break-down, as many as 30 to 45 years may have been gained (Velders et al., 2007). These estimates further underscore how early warning scientists must be attributed a role in curbing ODS-consumption being equally important to the Montreal protocol itself.

SCBA — social cost benefit analysis — to sweeten the pill (RMNO, 2008). The uptake of cost-benefit analysis has been most significant in the United Kingdom. And because the United Kingdom has promoted the extension of the method to EU regulations — with occasional support from other Member States — there are examples of formal requirements for cost benefit analysis in EU programmes. Findings from cost-benefit analysis are also referred to in impact assessments of new legislative proposals prepared by the Commission. There has been a rule in place for the past 15 years requiring a cost-benefit analysis as part of the screening of projects set to receive EU support under the Structural Fund programmes and a manual is available to guide these assessments, published by the European Commission (2008).

Obviously there are great methodological challenges in further expanding the use of cost-benefit analysis. However, there has also been some progress in these methodologies. For example, improvements in scientific knowledge and modelling techniques have helped to significantly influence the ratios of benefits to costs in favour of regulation.

Precautionary action can be justified by using credible estimates of the costs of inaction. The lead case illustrates that even if we have an understanding of only some of the benefits, making good use of the science base can be enough to prompt action. The nitrate case demonstrates that risk calculations can be useful and help prompt immediate action, even though the time-lag effects of exposure means that full proof will likely take decades to materialise. Finally, in the case of air pollution, making use of different estimates for mortality risk avoidance will help decision-makers to see that there are higher- and lower-bound estimates for the costs of inaction. Even if the lower-bound estimates are perhaps too conservative, with a bias towards health-effects, they will in many situations encourage more rather than less abatement effort. Reducing emission loads will also tend to bring relief for the intangible assets of biodiversity and nature.

Making the best use of environmental science and modelling helps to make environmental protection and precaution a priority. Producing cost estimates should not be left to economists alone, but should rather be seen as a starting point for a broader discussion, featuring also the relevant expertise in health, ecology, demography, modelling and science. Well-researched estimates, based on inter-disciplinary collaboration, can strengthen some of those scattered and diffuse interests, which during the ordinary processes of policymaking have difficulty making their voices heard.

References

ACCLPP, 2012, *Low level lead exposure harms children: A renewed call for primary prevention*, Report of the Advisory Committee on Childhood Lead Poisoning Prevention of the Centers for Disease Control and Prevention.

Ackerman, F., Massey, R., and Heinzerling, L., 2005, Applying cost-benefit to past decisions: Was environmental protection ever a good idea ?, *Georgetown Law Faculty Publications and Other Works*.

AEA Technology Environment, 2005, *Damages per tonne emission of $PM_{2.5}$, NH_3, SO_2, NO_X and VOCs from each EU25 member state (excl. Cyprus) and surrounding seas*, Service contract for carrying out cost-benefit analysis of air quality related issues in particular the Clean Air for Europe (CAFÉ) programme.

AMAP, 2011, Assessment of mercury in the Arctic, Oslo: Arctic monitoring and assessment programme.

Andersen, M.S., 2010, 'Miljøøkonomiske beregningspriser' (environmental economic unit prices), *Faglig Rapport 783*, Danmarks Miljøundersøgelser.

Andersen, M.S., Hansen, M.S., Carstensen, J., Kronvang, B., Andersen, H.E. and Thodsen, H., 2011, Monetary valuation with impact pathway analysis: Benefits of reducing nitrate leaching in European catchments, *International Review of Environmental and Resource Economics*, (5/3) 199–244.

ATSDR (Agency for Toxic Substances and Disease registry), 2011, Medical management guidelines: Sulfur Dioxide, Atlanta.

Axelrad, D.A., Bellinger, D.C., Ryan, L.M. and Woodruff, T.J., 2007, 'Dose-response relationships of pre-natal mercury exposure and IQ: An integrative analysis of epidemiologic data', *Environmental Health Perspectives*, (115/4) 609–615.

Barbosa, F., Tanus-Santos, J.E., Gerlach, R.F. and Parsons, P.J., 2005, 'A critical review of biomarkers used for monitoring human exposure to lead: Advantages, limitations and future needs', *Environmental Health Perspectives*, (113/12) 1 669–1 674.

Bell, M.L., Davis, D.L. and Fletcher, T., 2003, 'A retrospective assessment of mortality from the London smog episode of 1952: The tole of influenza and pollution', *Environmental Health Perspectives*, (112/1) 6–8.

Committee, 1997, *Japan's experience in the battle against air pollution: Working towards sustainable development,* Tokyo: Pollution-related health damage compensation and prevention association.

Diamond, P.A. and Hausman, J.A., 1994, Contingent valuation: is some number better than no number ?, *Journal of Economic Perspectives,* (8/4) 45–64.

De Roos, A.J., Ward, M.H., Lynch, C.F., Cantor, K.P., 2003, 'Nitrate in public water supplies and the risk of colon and rectum cancers', *Epidemiology, (*14) 640–649.

EEA, 2001, *Late lessons from early warnings: the precautionary principle 1896–2000*, Environmental issue report No 22, European Environment Agency.

European Commission, 2008, *Guide to cost-benefit analysis of investment projects,* Luxembourg.

Gidlow, D., 2004, 'In-depth review: Lead toxicity', *Occupational Medicine,* (54) 76–81.

Gould, E., 2009, 'Childhood lead poisoning: Conservative estimates of the social and economic benefits of lead hazard control', *Environmental Health Perspectives.*

Griffiths, C, McGartland, A and Miller, M, 2007, 'A comparison of the monetized impact of IQ decrements from mercury emissions', *Environmental Health Perspectives,* (115/6) 841–847.

Hammar, H. and Löfgren, Å., 2004, 'Leaded gasoline in Europe: Differences in timing and taxes', 192–205 in Harrington, W., Morgenstern, R. and Sterner, T. (eds), *Choosing environmental policy: Comparing instruments and outcomes in the United States and Europe*, Wash DC: RFF Press.

Heinzerling, L., Ackerman, F. and Massey, R., 2005, 'Applying cost-benefit to past decisions: was environmental protection ever a good idea?', *Admin Law Review,* (57) 155–192.

Hofstetter, P. and Hammitt, J.K., 2002, 'Selecting human health metrics for environmental decision-support tools', *Risk Analysis,* (22/5) 965–983.

Hylander, LD and Goodsite, ME, 2006, 'Environmental costs of mercury pollution', *Science of the Total Environment*, (368) 352–370.

IEA, 2009, *CO_2 emissions from fuel combustion — highlights,* International Energy Agency.

IPCC, 2007, *Safeguarding the ozone layer and the global climate system: Issues related to hydrofluorocarbons and perfluorocarbons*.

Japan Times Online, April 29, 2006, 'Koizumi issues official Minamata apology'.

Landrigan, P.J., 2002, 'The worldwide problem of lead in petrol', *Bulletin of the World Health Organization,* (80/10).

Lanphear, B.P., Hornung, R., Khoury, J., Yolton, K., Baghurst, P., Bellinger, D.C., Canfield, R.L., Dietrich, K.N., Bornschein, R., Greene, T., Rothenberg, S.J., Needleman, H.L., Schnaas, L., Wasserman, G., Graziano, J. and Roberts, R., 2005, 'Low-level environmental lead exposure and children's intellectual function: An international pooled analysis', *Environmental Health Perspectives,* (113/7) 894–899.

Menke, A., Muntner, P., Batuman, V., Silbergeld, E.K., and Guallar, E., 2006, 'Blood lead below 0.48 umol/L (10 ug/dL) and mortality among US adults', *Circulation,* (114) 1 388–1 394.

Meyer P., Brown M. and Falk, H., 2008, 'Global approach to reducing lead exposure and poisoning', *Mutation research,* (659/1–2) 166-175.

Miranda, M.L., Anthopolos, R. and Hastings, D., 2011, 'A geospatial analysis of the effects of aviation gasoline on childhood blood lead levels', *Environmental Health Perspectives*.

Molina, M.J. and Rowland, F.S., 1974, 'Stratospheric sink for chlorofluoromethanes: chlorine atom-catalysed destruction of ozone', *Nature,* (249/5460) 810–812.

Morgenstern, R., 1997, *Economic analyses at EPA*, Washington DC: Resources for the future.

NASA, 2010, *The world we avoided by protecting the ozone layer* (http:/earthobservatory.nasa.gov).

Needleman, H., 2004, 'Lead poisoning', *Annual Review of Medicine,* (55/1) 209–222.

Newman, P.A., Oman, L.D., Douglass, A.R., Fleming, E.L., Frith, S.M., Hurwitz, M.M., Kawa, S.R., Jackman, C.H., Krotkov, N.A., Nash, E.R., Nielsen, J.E., Pawson, S., Stolarski, R.S. and Velders, G.J.M., 2009, 'What would have happened to the ozone layer if chlorofluorocarbons (CFCs) had not been regulated', *Atmos. Chem. Phys,* (9) 2 113–2 128.

Nichols, A.L., 1997, 'Lead in gasoline', pp. 49–86 in R. Morgenstern (ed), *Economic analyses at EPA*, Washington DC: Resources for the future.

NSW (New South Wales) Government, 2006, Health factsheet methaemoglobinaemia (http://www.health.nsw.gov.au/factsheets/general/methae_fs.html).

NYT, 2011, 'EPA plans to revisit a touchy topic — the value of saved lives', 18 January 2011, *New York Times.*

OECD, 1977, *Environmental Performance Review: Japan*, Paris.

OECD, 2006, *Cost-benefit analysis and the environment: recent developments*, Paris.

OECD, 2008, *Costs of inaction on key environmental challenges*, Paris.

Olson, K., 2004, *Poisoning and drug overdose,* 4th ed., McGraw-Hill Medical.

Opinion, 1977, 'Million-dollar problem—billion-dollar solution?', *Nature,* (268/5616) 89.

Pizzol, M., Thomsen, M., Frohn, L.M. and Andersen, M.S., 2010, 'External costs of atmospheric Pb emissions: valuation of neurotoxic impacts due to inhalation', *Environmental Health,* (9) 9.

Pope, C.A., Thun, M.J., Namboodiri, M.M., Dockery, D.W., Evans, J.S., Speizer, F.E. and Heath Jr., C.W., 1995, 'Particulate air pollution as a predictor of mortality in a prospective study of U.S. adults', *American Journal of Respiratory and Critical Care Medicine,* (151/3) 669–674.

Pope C.A., Burnett, R.T., Thun, M.J., Calle, E.E., Krewski, D., Ito, K. and Thurston, G.D., 2002, 'Lung cancer, cardiopulmonary mortality and long-term Exposure to fine particulate air pollution', *JAMA,* (287/9) 1 132–1 141.

Pretty, J., Brett, C., Gee, D., Hine, R., Mason, C., Morison, J., Rayment, M., van der Bijl, G. and Dobbs, T., 2001, 'Policy challenges and priorities for internalizing the externalities for modern agriculture', *Journal of Environmental Management and Planning,* (44/2) 263–283.

Reddy, A. and Braun, C.L., 'Lead and the Romans', *Journal of Chemical Education,* (87/10) 1 052–1 055.

Rive, N., 2010, 'Climate policy in western Europe and avoided costs of air pollution control', *Economic Modelling,* (27/1) 103–115.

RMNO, 2008, *Social cost-benefit analysis for environmental policy-making, Background study V14*, Raad voor ruimtelijk, milieu- en natuuronderzoek, The Hague.

Roan, S., 1989, *Ozone crisis: The 15 year evolution of a sudden global emergency*, New York: John Wiley.

Salkever, D.S., 1995, 'Updated estimates of earnings benefits from reduced exposure of children to environmental lead', *Environmental Research,* (70/1) 1–6.

Schnaas, L., Rothenberg, S.J., Flores M.F., Martinez, S., Hernandez, C., Osorio, E., Velasco, S.R. and Perroni, E., 2006, 'Reduced intellectual development in children with prenatal lead exposure', *Environmental Health Perspectives,* (114/5) 791–797.

Schwartz, J., 1994, 'Societal benefits of reducing lead exposure', *Environmental Research,* (66/1) 105–124.

Shimshack, J.P. and Ward, M.B., 2010, 'Mercury advisories and household health trade-offs', *Journal of Health Economics,* (29/5) 674–685.

Slaper, H., Velders, G.J.M. and Matthijsen, J., 1998, 'Ozone depletion and skin cancer incidence: a source risk approach', *Journal of Hazardous Materials,* (61/1–3) 77–84.

Smith, K.R., Jerrett, M., Anderson, H.R., Burnett, R.T., Stone, V.,Derwent, R., Atkinson, R.W., Cohen, A., Shonkoff, S.B., Krewski, D., Pope, C.A., Thun, M.J. and Thurston, G., 2009, 'Public health benefits of strategies to reduce greenhouse-gas emissions: Health implications of short-lived greenhouse pollutants', *The Lancet,* (374/9707) 2 091–2 103.

Spadaro, J.V. and Rabl, A., 2004, 'Pathway analysis for population-total health impacts of toxic metal emissions', *Risk Analysis,* (24/5) 1 121–1 141.

Spadaro, J.V. and Rabl, A., 2008, 'Global health impacts and costs due to mercury emissions', *Risk Analysis, (*28/3) 603–613.

Stern, N., 2007, *Stern review on the economics of climate change*, London: HM Treasury.

Sunstein, C.R., 2007, 'Of Montreal and Kyoto: a tale of two protocols', *Harvard Environmental Law Review,* (31) 1–65.

Taylor, M.P., Schniering, C.A., Lanphear, B.P. and Jones, A.L., 2010, 'Lessons learned on lead poisoning in children; Onehundred years on from Turner's declaration', *Journal of Paediatrics and Child Health,* (1) 440–1 754.

UNEP 2009a, *Key achievements of the Montreal Protocol to Date*, United Nations Environment Programme, Nairobi.

UNEP 2009b, 'Ozone Treaty press release', 16 September 2009, United Nations Environment Programme.

US EPA, 1985, Costs and benefits of reducing lead in gasoline: Final regulatory impact analysis, United States Environmental Protection Agency, National Center for Environmental Economics, Washington DC.

US EPA, 2010, Valuing mortality risk reductions for environmental policy: A white paper (draft), United States Environmental Protection Agency, National Center for Environmental Economics, Washington DC.

Van Grinsven, H.J.M., Rabl, A., de Kok, T.M., 2010, 'Estimation of incidence and social cost of colon cancer due to nitrate in drinking water in the EU: a tentative cost-benefit assessment', *Environmental Health,* (9) 58.

Velders, G.J.M., Slaper, H., Pearce, D.W. and Howarth, A., 2001, *Technical report on stratospheric ozone depletion*, RIVM report 481505011, Bilthoven.

Velders, G.J.M., Andersen, S.O., Daniel, J.S., Fahey, D.W. and McFarland, M., 2007, 'The importance of the Montreal protocol in protecting climate', *Proceedings of the National Academy of Sciences in the United States of America,* (104/12) 4 814–4 819.

Von Bahr, J. and Janson, J., 2004, *Cost of late action — the case of PCB*, TemaNord 556, Nordic Council of Ministers.

Weyer, P.J., Cerhan, J.R., Kross, B.C., Hallberg, G.R., Kantamneni, J., Breuer, G., Jones, M.P., Wei, Z. and Lynch, C.F., 2001, 'Municipal drinking water nitrate level and cancer risk in older women: The Iowa women's health study', *Epidemiology,* (12/3) 327–338.

WHO, 1994, Health and environmental effects of ultraviolet radiation; A Scientific Summary of Environmental Health Criteria, World Health Organization.

WHO, 2007, Nitrate and nitrite in drinking water, *Background document for development of WHO guidelines for drinking water quality*, WHO/SDE/WSH/07.01/16, World Health Organization.

World Bank, 2007, *Cost of pollution in China: Economic estimates of physical damages*, Washington DC.