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COSTS OF INACTION ON ENVIRONMENTAL POLICY CHALLENGES

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FOREWORD

When they met in April 2004, OECD Environment Ministers drew attention to the need for more analysis of the "costs of inaction" (COI) on key environmental challenges. Ministers also asked the OECD to work on this theme, and to report back at their next meeting. This report is part of the response to that request.

A high-level meeting of the OECD Environment Policy Committee was held (April 2005), to launch discussion on this topic. Since then, the OECD effort has concentrated on preparing final reports to respond to the mandate given by Environment Ministers in 2004. This report provides technical information on the costs of inaction in selected environmental policy areas. The report is not comprehensive – it covers neither all environmental problems nor all dimensions of those environmental problems it does address. The aim of the report is merely to offer some introductory perspectives on the topic; to provide some initial views based on the current literature; and to suggest some of the problems likely to be encountered in moving further in this (highly complex) area.

The report was drafted by Nick Johnstone, Ivan Hascic, and Tom Jones of the OECD Environment Directorate, working under the guidance of the OECD Environment Policy Committee and its Working Party on National Environmental Policies. It has also been materially improved by comments received along the way from delegates in OECD capitals and from other members of the Secretariat.

The report is published under the responsibility of the OECD Secretary-General.

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EXECUTIVE SUMMARY

When they met in April 2004, OECD Environment Ministers drew attention to the need for more analysis of the "costs of inaction" (COI) on key environmental challenges. This report is part of the response to that request. It is important to be clear at the outset about what is meant by the terms *cost* and *inaction*. OECD countries have made significant strides in addressing many of the environmental concerns discussed in this report. The term "inaction" must therefore be interpreted in this context. While the continued implementation of existing regulatory and market-based policy instruments at their existing level of stringency can hardly be characterised as "inaction" in a strict sense, adopting such a perspective is likely to be more instructive (and easier to apply) than ignoring the existing policy framework. As such, this report uses an assumption of "no new policies beyond those which currently exist" as the basis for its analysis of "inaction".

With respect to "costs", both market and non-market impacts are considered in much of the literature reviewed in this report. This includes the *direct financial costs* of inaction associated with environmental degradation, such as expenditures on remediation and restoration, private and public health services costs, and private defensive expenditures. Other more *indirect costs* include the costs of resource depletion and environmental degradation which are reflected in other associated markets (*i.e.* real estate and labour markets), as well as general equilibrium impacts.[1] In addition, costs associated with the loss of *environmental use values* which are not reflected in markets at all must be included. This includes non-market costs associated with pain and suffering, and some aspects of environmental quality (aesthetics, visibility, *etc.*) And finally, a full estimate of the costs should reflect *non-use values*, such as existence values associated with biodiversity, as well as values associated with bequest and altruism.

When valuing the "costs of inaction", there are several methodological issues that need to be considered:

- Uncertainties with regard to both environmental impacts and the economic value of those impacts (including uncertainty about technological trajectories over time);
- Thresholds and irreversibilities, which can lead to "discontinuous" impact functions;
- The long-run nature of environmental problems (and thus the need for "discounting" the streams of anticipated costs);
- The degree of substitutability between environmental resources and other inputs into the economy;
- The distribution of environmental impacts, and their links to social concerns about equity; and
- The endogeneity of responses to changing environmental conditions (*e.g.* adaptation).

[1] For instance, in the valuation of public service health costs, it is important to take into account the means by which that service is financed. If it is financed through general tax receipts, the costs of inaction will be greater, the more distortionary the existing system of taxation.

Despite these complexities associated with valuation, the literature reviewed for this report suggests that the economic costs of failing to introduce environmental policies, or of introducing policies that are not "sufficiently ambitious"[2], can be considerable. For example:

- *Air pollution* can lead to reduced agricultural yields, degradation of physical capital, and broader impacts on ecosystem health. The costs of not introducing the EC's "Thematic Strategy on Air Pollution" are estimated to represent about 0.35-1.0% of EU-25 GDP in 2020 (EC, 2005). Although some of the tangible health costs of pollution (lost productivity, health service costs, *etc.*) may be more visible, economic studies suggest that more intangible costs, such as "pain and suffering", are very significant as well.

- In non-OECD countries, the economic impacts of inaction with respect to *water pollution* may be even greater. According to the WHO (Prüss-Üstün *et al.*, 2004), 1.7 million deaths and 4.4% of the burden of disease (BoD)[3] are attributable to unsafe water supply, sanitation and hygiene (WSH). Ninety per cent of the deaths involve children under 5 years old. Households devote significant resources (time and money) to securing access to clean water, in order to these health impacts.

- Estimates of the economic costs of *climate change* vary widely, with recent assessments generating figures as high as 14.4% in terms of per capita consumption equivalents (Stern, 2007a)[4], when both market and non-market impacts are included. While there is significant uncertainty about the eventual costs of inaction with respect to climate change, few would doubt that it has the potential to have very significant implications for the world economy – particularly in non-OECD countries. Reduced agricultural yields, increased sea-level rise, and greater prevalence of some infectious diseases are likely to significantly disrupt these economies.

- Environment-related *industrial hazards* – such as oil spills and land contamination – are already generating significant costs of inaction. For example, experience in Europe and US nevertheless indicates that the costs of remediation of damaged ecosystems can run into billions of Euros. Moreover, due to the irreversible nature of some of the impacts that can be expected, the costs of restoration or remediation (no matter how comprehensive) will only represent a proportion of the total social costs of inaction.

- While the economic risks associated with *natural disasters* (*e.g.* floods, hurricanes) are only partly attributable to environmental factors, and can only be partly reduced through public policy measures (*e.g.* mitigation of climate change, flood prevention measures), the costs of inaction in these areas can also be considerable – the World Bank (2006) has estimated that the costs of natural disasters for the poorest countries can be as much as 13% of annual GDP.

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2 In economic terms, this includes policies whose further strengthening would generate marginal benefits in excess of marginal costs. However, as noted below, this paper does not assess the costs of policy interventions (*i.e.* benefits of inaction).

3 BOD is measured in terms of disability-adjusted life years (DALYs) – a common indicator used in cost-effectiveness studies in the health economics field.

4 The metric used in Stern (2007), which has caused some confusion, is an attempt to express a complex issue in a concise manner. Assuming future growth rates in the absence of any economic impacts from climate change, the consumption path associated with that growth rate is first calculated. Next, climate change impacts are considered, which are translated into lower future growth rates, and a correspondingly lower future consumption path. The cost of inaction is thus the difference between these two consumption trajectories [see Sterner and Persson (2007) for clarification].

- The costs of unsustainable *natural resource management*[5] – in terms of lost future benefits from resource exploitation – can be considerable. For example, Bjørndal and Braso (2005) concluded that inefficient management of the east Atlantic bluefin tuna fishery may be resulting in reduced fishery yields with a discounted value of USD 1-3 billion. However, the costs of unsustainable fisheries management extend well beyond these direct impacts on the resources themselves, to also include indirect impacts on "downstream" sectors and ecosystems.

These results should, however, be interpreted with caution. Given the uncertainties, as well as the fundamental methodological difficulties associated with estimating the costs of inaction, it would be foolhardy to attempt to "cost" environmental policy inaction in any aggregate sense. However, it is clear that there are many environmental problems for which the costs of not taking further policy action *are* significant – and are already directly affecting OECD economies in a variety of ways.

It is also important to realize that some of these costs are already being reflected in household budgets and firms' balance sheets. Increased costs are incurred in an effort to secure access to increasingly scarce resources, and "defensive" expenditures are incurred in order to avoid the impacts of environmental degradation. For example, expenditures incurred to secure access to clean water in developing countries can be a very significant proportion of a household's budget.

Some of the financial costs of environmental policy inaction are also already being reflected directly in public sector budgets – *i.e.* increased public expenditures on health services due to air and water pollution, unemployment benefits and adjustment programmes for out-of-work fishers, remediation costs for contaminated sites, dikes and other measures to protect against flooding and extreme weather events, *etc.* Thus, many of the costs of environmental policy inaction are already reflected in a diffuse manner throughout the government's balance sheet.

Other components of the costs of inaction may be reflected (at least in part) in existing markets, even though they are not readily perceived as costs of environmental policy inaction *per se*. Examples include the effects of contaminated sites on adjacent property prices, the effects of air pollution on agricultural yields, or the cost of property insurance in coastal areas. All of these costs are attributable in part to environmental policy inaction.

The impacts of other elements of the costs of environmental policy inaction may not be reflected in economic variables in an identifiable manner. For example, the costs associated with the continued loss of marine and terrestrial biodiversity are likely to be very significant, but their impacts are not reflected in market prices or national accounts in an identifiable manner. This is also the case with other more intangible and subjective aspects of the costs of inaction, such as "pain and suffering" from ill-health. These impacts may impose a very significant burden associated with "inaction" (in terms of lost welfare), so they should not be neglected.

Thus, while there is significant economic and scientific uncertainty associated with the estimates in different areas, there is little question that for a number of areas such costs are already significant, affecting many markets and sectors, as well as important macroeconomic variables. Put another way, inadequately stringent environmental policies in some areas can serve as a significant brake on economic productivity and growth.

5 Fisheries and groundwater abstraction were selected for review in this report. While undoubtedly important, the issue of biodiversity is not addressed directly. However, many of the areas reviewed (fisheries, climate change, air and water pollution) have direct implications for biodiversity.

However, even if the costs of inaction are deemed to be significant, identifying those areas in which existing environmental policies should be strengthened or new environmental policy initiatives undertaken would still require a careful balancing of the marginal costs of inaction with the marginal costs of further reducing the associated impacts beyond those measures already in place. This report does not review the (vast) literature on the costs of *action*. In the absence of information about the costs of policy interventions, estimates of the (marginal) costs of inaction on their own cannot be considered as a guide to either the establishment of policy priorities or to overall economic efficiency.

CHAPTER 1. INTRODUCTION

When they met in April 2004, OECD Environment Ministers drew attention to the need for more analysis of the "costs of inaction" (COI) on key environmental challenges. This report is part of the response to that request.

It begins with an overview (Chapter 1) of some of the definitional questions that underlie the notion of "costs of inaction", as well as some of the methodological issues that this concept embodies. Chapters 2-5 then summarise the COI literature dealing with selected environmental challenges (Chapter 2 on air and water pollution; Chapter 3 on climate change; Chapter 4 on environmental hazards, accidents, and natural disasters; and Chapter 5 on natural resource management). A few conclusions are offered at the end of the report (Chapter 6).

Defining what is meant by the "costs of inaction" is not straightforward. This paper uses an assumption of "no new policies beyond those which currently exist" as the basis for its discussion of "inaction". With respect to "costs" both market and non-market impacts are considered, but in some cases it is difficult to obtain reliable estimates of the costs of environmental impacts that are not reflected (directly or indirectly) in market prices and national accounts.

Some of these costs are already reflected in household budgets and firms' balance sheets (*e.g.* additional costs to secure increasingly scarce resources; or "defensive" expenditures, aimed at avoiding the impacts of environmental degradation). Similarly, some of the financial costs of inaction are also already reflected directly in public sector budgets (*e.g.* increased public expenditures on health services due to air and water pollution; or cleanup costs at contaminated sites). Other costs may be reflected (at least in part) in existing markets, even though they are not readily perceived as costs of environmental policy inaction *per se* (*e.g.* the effects of environmental degradation on adjacent property prices; or the additional cost of flood insurance in coastal areas). Looking beyond these market-based costs, there are also costs of inaction associated with a wide range of intangibles (*e.g.* pain and suffering from being in ill-health) and various forms of ecosystem degradation. Although less visible, these non-market costs are likely to be important.

It is important to emphasise that this report does not review the entire (and vast) body of literature on the costs of *inaction*. It is necessarily selective, focusing on a few areas. [6] Moreover, no attempt is made to review the equally vast body of literature on the costs of *action*. In the absence of information about the costs of policy interventions, estimates of the (marginal) costs of inaction on their own cannot be considered as a guide to either the establishment of policy priorities or to overall economic efficiency.

In addition, much of the information presented here is expressed in terms of *total* costs – not *marginal* costs. Although it is only the introduction of policies whose *marginal* benefits are expected to exceed their *marginal* costs that will increase economic efficiency, the underlying premise of this report is that information about the *total* costs of inaction is still of interest, in the sense that it provides a broad indication of the scale of the costs of inaction in various fields of environmental policy.

[6] For example, the cost of policy inaction with respect to biodiversity is only addressed insofar as such impacts arise out of policy inaction in other areas (*e.g.* fisheries management, climate change) which are addressed in the report.

What do we mean by "costs of inaction"?

Defining "inaction"

All OECD governments have already introduced policies to conserve scarce natural resources and/or preserve environmental quality. Defining policy "inaction" in the context of an area of public policy in which significant strides have already been made is clearly not straightforward. Conceptually, there are at least three possible baselines that could be used to represent "inaction":

- a hypothetical scenario, in which it is assumed that there is *no environmental policy intervention whatsoever*;
- an assumption that *existing environmental policy continues* in its present form and at its present level of stringency; and
- an assumption that credible commitments will be implemented that would *increase the level of environmental policy ambition in the future*.

In specific circumstances, it may be appropriate (and possible) to define some absolute notion of "inaction" along the lines set out in the first bullet above. For example, for a very recently-discovered social "bad", the relevant notion of inaction may be one in which there is no existing policy framework at all. Indeed, the most common use of the term "costs of inaction" used in contemporary policy debates relates to the onset of HIV in developing countries, and this is the notion of "inaction" which is generally applied there (*e.g.* World Bank, 2003).

An analogous situation in the environmental domain might be the discovery of the hole in the ozone layer – a problem that arose from the emission of chlorofluorocarbons and other ozone-depleting substances. When initially discovered in the 1970s, the most appropriate definition of "inaction" may well have been a situation in which there was no relevant policy framework at all in place. No actions had yet been taken to reduce emissions of ozone-depleting substances, except for endogenous actions (by firms), motivated by efforts to reduce production costs or to improve product quality in general.

Arguably, in the work undertaken by the World Bank in the mid–1990s on environmental degradation in developing countries, a similar perspective was adopted – in the sense that this work assessed environmental impacts in countries in which an environmental policy framework was in its very early stages of development. In this context, the damages from "inaction" with respect to air and water pollution in China were reported to have amounted to almost 8% of GDP in 1995 (World Bank, 1997). Similarly, the annual damage cost of environmental degradation in 2000 in Lebanon was estimated to be 3.4% of GDP (close to USD 5665 million per year). The figure for Tunisia was 2.1% of GDP (nearly USD 440 million in 1999) (World Bank, 2004a).

The second possibility (bullet 2 above) is to start from the "existing policy framework". Thus, a series of studies undertaken by the European Commission on the "costs of non-Europe" (commonly referred to as the "Cecchini Report") (EC, 1998) estimated the cost saving to the European Union economies of removing internal frontier controls within the Union (consisting of 12 Member States at the time) at EUR 8 billion. At the time this report was completed, there had already been considerable economic integration among countries, so any approach that ignored this "existing integration" would clearly have been less informative than one which took it into account.

As already mentioned, OECD governments have introduced a wide variety of policies, aimed at preserving environmental quality or conserving natural resources. The continued implementation of these regulations and market-based policies at their existing level of stringency can hardly be characterised as "inaction" in a strict sense, and is perhaps better characterised as "business–as–usual". Nonetheless, adopting such a perspective is likely to be more instructive (and easier to apply) than "assuming away" the existing policy framework.

The third possible definition of "inaction" (bullet 3 above) would involve incorporation of existing commitments to policy reform – commitments which go beyond the existing policy framework. For instance, "inaction" might be assumed to be based upon the commitments that some countries have previously agreed to under the Kyoto Protocol, with respect to climate change – or under the Millennium Development Goals, with respect to water supply and sanitation. Although many countries are likely to fall short of fully achieving these commitments with existing policies, it may be deemed appropriate to adopt a dynamic perspective of policy development – a perspective in which it is assumed that efforts will be made to meet these commitments in future.

The perspective taken in this report is that the most practical and informative perspective to adopt for "inaction" is one in which it is assumed that the existing policy regime (*i.e.* the *status quo*) is kept in place (*i.e.* bullet 2 above). This is consistent with the methodology adopted in the OECD *Environmental Outlook to 2030* (OECD, 2008), in which the baseline modelling scenario assumes that "currently existing policies are maintained, but no new policies are introduced to protect the environment." This has the pragmatic advantage that it gives governments "credit" for actions they have already taken, but not for those they have simply promised (and may never achieve). However, using this definition immediately raises the question of what exactly is embodied in the *status quo* policy framework, and how this can best be modelled in a dynamic context.

Once the "baseline" policy scenario has been defined in general terms, it is then important to assess how economic agents are likely to *respond* dynamically to that scenario. This response will depend in part upon the nature of the policy instrument(s) being implemented within the existing policy. For instance, the retention of an existing cap for tradable emission permits will have very different implications for the costs of inaction than the retention of a given set of performance standards for the same pollutants – even if the underlying environmental objective is the same. A cap and trade system that limits emissions will be unaffected by economic growth rates, firm entry (and exit), and technological innovation. This will not be true of performance standards – at least not without continuous adjustment of the policy measure. Over time, therefore, different policy measures will involve different *scale* and *substitution* effects – both of which will eventually translate into a different shape and location of the "costs of inaction" pathway.

Households and firms are also likely to respond to the changing environmental conditions that they face, and the nature of this adaptation to the state of the existing environment should be reflected in the analysis. It cannot realistically be assumed that those who will be affected by a degrading environment will not adjust their behaviour in the face of that degradation. A corollary applies for cases involving local environmental "bads" (*e.g.* hazardous waste facilities or local air pollution), but where *private* markets are affected by *public* environmental conditions. As environmental conditions change, associated (private) markets will be affected (*e.g.* real estate) and households will adjust. All of this will again affect the shape and location of the costs of inaction.

Defining "costs"

There will also be residual environmental consequences embedded in the "no new policies" assumption that has just been described. There are several different units (or metrics) in which these environmental consequences can be expressed, but the broadest distinction that can be made is between "physical" (ecological, health, *etc.*) metrics and "monetary" ones. Metrics related to resource exploitation might include measures such as rates of deforestation, rates of water abstraction relative to availability, and assessments of the status of fish stocks. Metrics related to environmental degradation might include emission rates relative to assimilative capacity. Further downstream, impacts on such variables as health, material damages, and resource productivity may also need to be assessed.

The standard procedure for assessing environmental impacts is environmental impact analysis (EIA). In the context of assessment of the costs of inaction, an EIA would measure the various environmental impacts in physical units (which will probably vary from one impact to another). A life-cycle analysis (LCA) amounts to performing an "extended EIA", with environmental impacts being measured across the entire life cycle of the environmental problem in question.

Taking the additional step of estimating the value of these impacts in monetary terms would then lead to two key advantages:

- Different types of impacts associated with inaction can then be compared using a common metric (*i.e.* loss of biodiversity and human health impacts);
- The estimated costs of *inaction* can then be directly compared with the costs of *action* (*i.e.* the benefits of inaction, such as avoided investment and other costs).

However, actually taking this "valuation step" is not easy, mainly because many environmental damages relate to impacts that do not have a market value. Whether because of the existence of externalities or the absence of enforceable property rights, there may be no financial cost associated with resource depletion or environmental quality degradation. Even if there is a market value, this value may not reflect the real economic value: for example, the price of fish in the market may not reflect scarcity rents associated with its capture; the investment and operating costs associated with wastewater treatment plants may not reflect the full social costs associated with pollution.

Figure 1 illustrates one way of thinking about this problem. In the innermost circle, the *direct financial costs* of inaction associated with environmental degradation are captured. This might include expenditures on remediation and restoration, private and public health services costs, and private defensive expenditures. Proceeding outward to the next bubble, other more *indirect costs* are included. These capture some of the indirect costs of resource depletion and environmental degradation which are reflected in other associated markets (*i.e.* real estate and labour markets), as well as general equilibrium impacts.[7] In the next bubble, costs associated with the loss of *environmental use values* which are not reflected in markets at all are included. This would include the non-market costs associated with "pain and suffering", as well as some aspects of environmental quality (aesthetics, visibility, *etc.*) And finally, the last bubble incorporates the loss of *non-use values*, such as existence values, as well as values associated with bequest and altruism.

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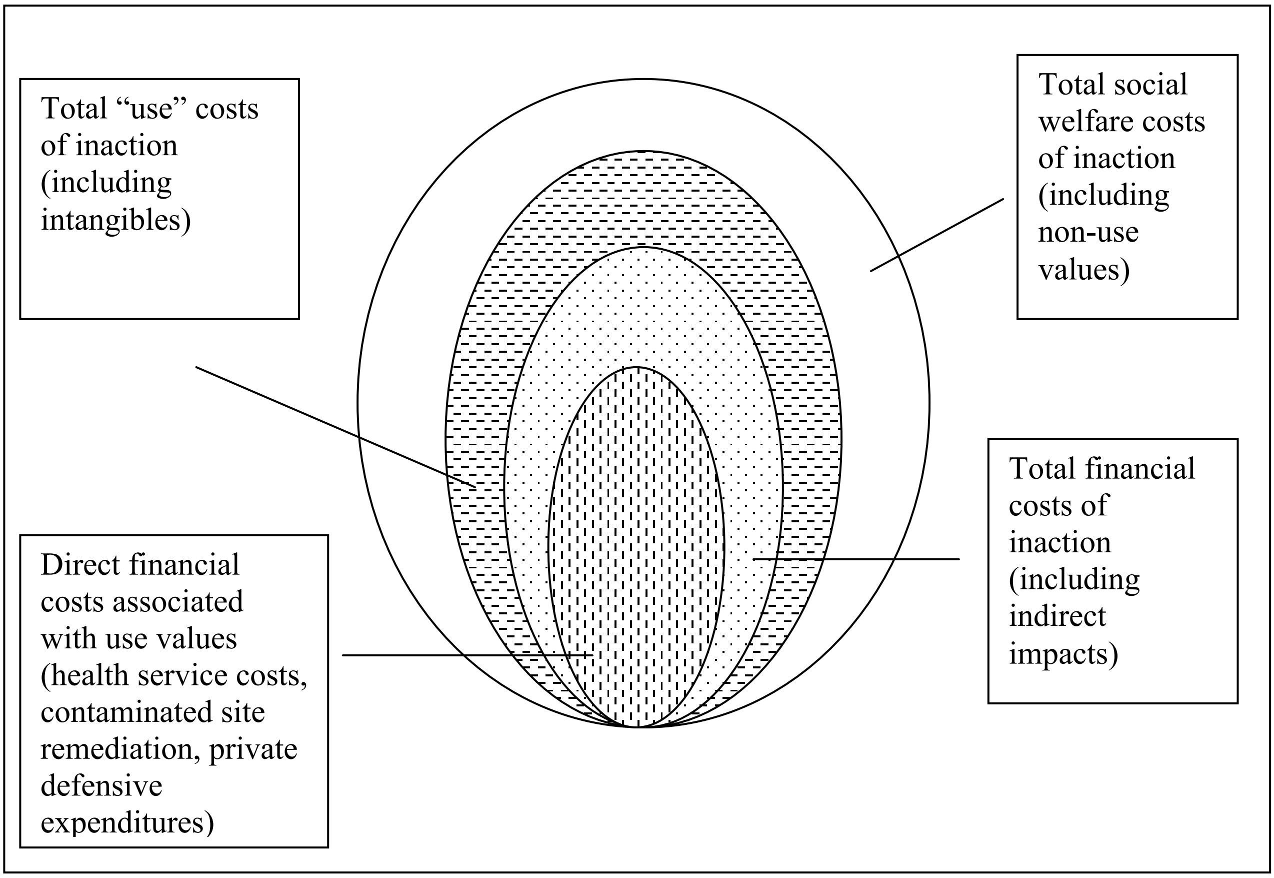
7 For instance, in the valuation of public service health costs, it is important to take into account the means by which that service is financed. If it is financed through general tax receipts, the costs of inaction will be greater, the more distortionary the existing system of taxation.

Figure 1. Unbundling the Costs of Inaction



Estimates of the costs of inaction should, in principle, reflect all of these values. Two broad approaches have been developed to resolve the problem of placing a value on environmental assets: (i) revealed preferences; and (ii) stated preferences. In the case of *revealed* preferences, efforts are made to derive the value of environmental assets from behaviour in existing markets for "associated" goods and services. For instance, the cost of polluted air may be reflected indirectly in real estate markets. Alternatively, efforts to value environmental assets through *stated* preference techniques posit a hypothetical market, for which respondents are requested to value changes in environmental conditions directly.

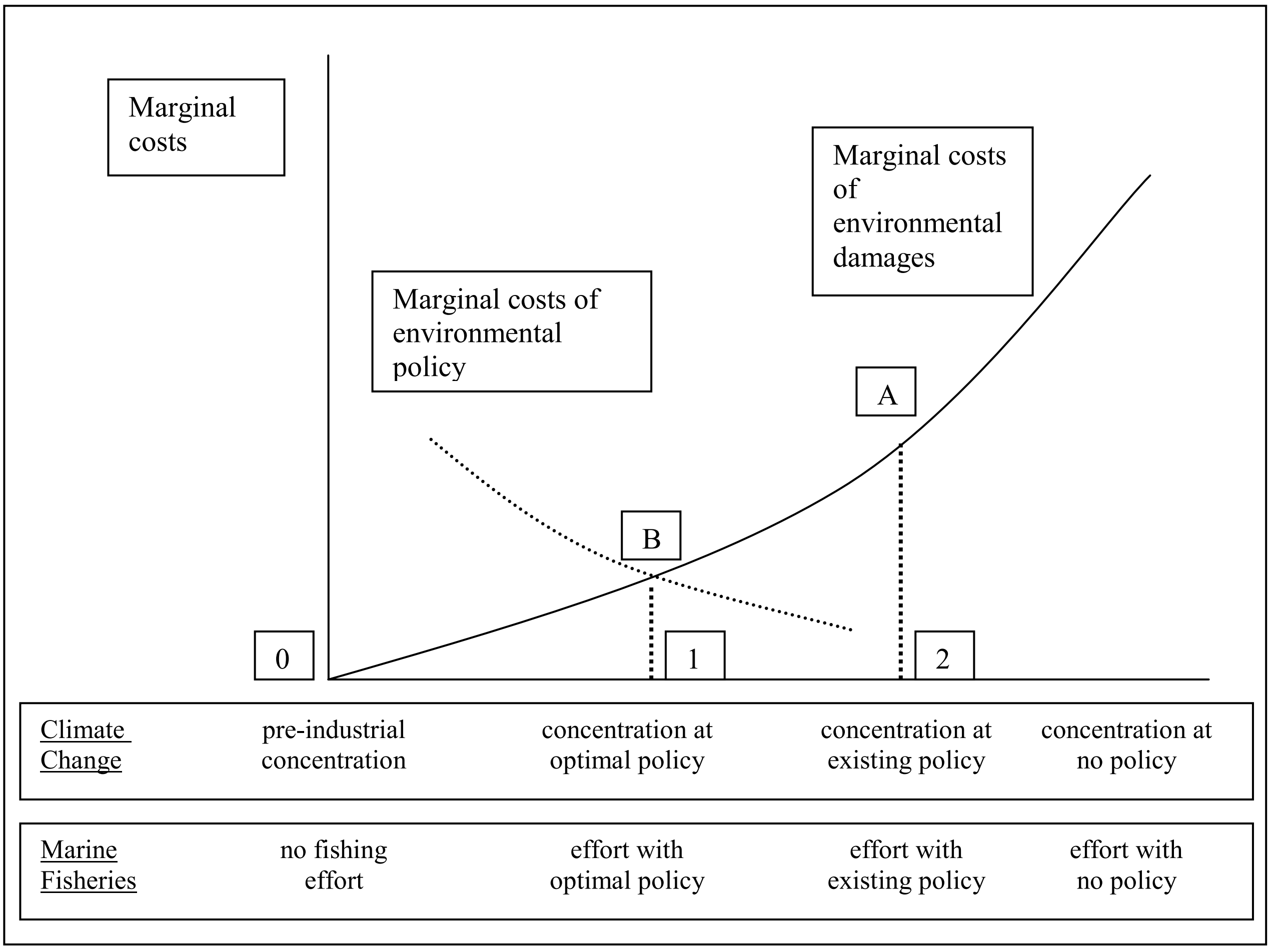
Putting it all together: Assessing the total costs of inaction

Figure 2 integrates the main elements of the previous discussion, drawing upon the climate change and fisheries examples.[8] "Inaction" has been defined in terms of the "continuation of existing policies". This level of "inaction" will result in a given concentration of greenhouse gases or fish stock at a specific point in time (Line 2 in Figure 2).[9]

8 The figure is the "mirror" image of the more usual representation of policy costs and benefits in which "effort" (*e.g.* abatement or conservation) is increasing from left to right on the x-axis. However, since the focus of this report is on "inaction", the x-axis is reversed.

9 In the case of fisheries, this is an over-simplified representation. Ideally, fisheries policies should target fish stocks, and there is no one-to-one relationship between fishing effort (or "total allowable catch") and

Figure 2. Marginal and Total Costs of Inaction



The marginal costs of environmental damages increase as the ambition level of policy declines (*i.e.* "inaction" increases). This curve intersects with the line representing policy inaction at Point A – which can be interpreted as the *marginal* costs of inaction. Conversely, the marginal costs of *addressing* the environmental problem rise with the level of policy ambition. In the graph, this is represented as the "decreasing marginal costs of environmental policy" curve, since the level of policy stringency decreases as one moves to the right on the x-axis.[10]

The efficient level of policy stringency is the point where the marginal costs of environmental damages and marginal costs of associated policy interventions intersect (Point B). This level of policy stringency serves as one possible counterfactual to "inaction", and is assumed here to be more stringent than current policies. A second possible counterfactual would reflect a level of stringency involving no anthropogenic contributions to environmental degradation or resource exploitation (point 0).

the fish stock at a given point in time. This is because the latter will also depend upon past harvest rates and ecological conditions.

10 While the hypothetical curves presented here are continuous, there may actually be important discontinuities (*e.g.* thresholds and irreversibilities), resulting in major changes in estimated impacts (and in the estimated "costs of inaction").

To arrive at a value for the *total* costs of inaction, it is necessary to calculate the area under the "marginal costs of environmental damages" curve, between the points representing "inaction" and the assumed counterfactual. If the comparison is made with the *optimal* level of policy intervention, the total costs of inaction will be represented by the area 1BA2; if the comparison is made with the highest possible level of stringency, the total costs of inaction will be represented by the area OA2.

Criteria for selecting the issues examined in this report

It is not possible here to provide an exhaustive review of the literature on the costs of inaction with respect to *all* environmental policies. The particular issues that are addressed in this report (health impacts of pollution, climate change, environment-related accidents, hazards and disasters, and natural resource management) have been selected for discussion for two main reasons: (i) taken together, they provide examples of the most common issues which arise when measuring the costs of inaction; and (ii) they represent particular environmental problems for which political pressure related to "inaction" seems most likely to occur.

For instance, one of the most controversial issues relates to the values that are placed on non-use values (or passive use values). These values can only be estimated using stated preference techniques, because they do not "leave a behavioural trail" (Pearce *et al.* 2006). Obtaining reliable estimates of these impacts will require significant care in eliciting and analysing the preferences of the respondents. One particularly controversial area is the notion of "existence" values – an example of which is the value which respondents place on species preservation, even though they may never derive use benefit from the continued existence of that species.[11] Non-use values are especially relevant for the problem of inaction in the field of *natural resource management*, where impacts on ecosystems and biodiversity can be significant.[12]

Even some use values can be controversial to value. The area of human health is one such case. Estimating the "costs of illness", such as hospital admission costs, medicine costs, and lost productivity is relatively straightforward, at least in principle. However, this will not encompass all the negative impacts associated with health degradation, since important intangible costs (*e.g.* pain and suffering) will be ignored. Even more controversial is the estimation of the value of mortality, as reflected in the estimated value of a statistical life.[13] For both reasons, the inclusion of a discussion of policy inaction related to *health impacts caused by air and water pollution* was seen to be of interest for this report.

Dealing with the very long run adds an additional level of complexity to the "costs of inaction" problem. Carbon dioxide emitted today has an atmospheric lifetime of over 200 years; air pollutants to which people are exposed today can generate adverse health impacts in 50-60 years; over-exploited fish stocks can take decades to recover. Costs today also have a higher value than those borne in the future, both because of "pure time preference" and the "opportunity cost of capital". The further into the future the cost occurs, the lower the weight that will tend to be attached to it. Indeed, the estimated present value of the costs of inaction can vary by orders of magnitude, with even small changes in the discount rate that is applied. Some people even find the practice of discounting morally unacceptable, because it seems to suggest that future costs are less important than present ones - and is therefore unfair to future generations. Temporal

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11 "Existence" values are often mistaken for "intrinsic" values. The latter are unrelated to human preferences.

12 The costs of inaction associated with biodiversity are indirectly addressed in this report, via the discussions on groundwater depletion and fisheries management.

13 Objections to estimating the value of a statistical life are common on ethical grounds. However, casual inspection indicates that people will not allocate an infinite amount of resources to reduce a marginal change in risk.

considerations such as these lie at the heart of the *climate change* and *natural resource management* issues, so this is another dimension on which this report focuses.

Environmental pressures can also embody complicated non-linear impacts, so any focus on the costs of inaction should embody a closer examination of some of the dynamic issues involved with this non-linearity. Three issues seem to be important in this regard:

- *Cumulative effects:* Some environmental impacts will become significantly greater as a result of cumulative environmental pressures over time. Many *health-related impacts* exhibit such an effect, such as bio-accumulation of hazardous substances in the food chain.
- *Thresholds:* There are numerous areas in which impacts may increase sharply, once a particular level (threshold) of environmental pressure is exceeded. In the area of *climate change*, thermohaline circulation is one example – in effect, there may be a "tipping point", after which an inversion might arise (with significant implications for the total costs of inaction).
- *Irreversibilities:* While some environmental impacts are potentially "reversible" (allowing for the restoration of environmental conditions to their prior state), there are many areas in which this is not the case (once degraded, environmental values are lost permanently). Species loss associated with unsustainable *natural resource management* and *environment-related hazards*, such as soil contamination provide two examples here.

In the presence of such non-linearities, the costs of preventing environmental degradation in the first place (mitigation) will often be less than the costs of addressing the impacts of the environmental problem once it has occurred (restoration). Indeed, for many types of impacts - particularly for those involving irreversibilities - it is not possible at all to restore the environment to its previous state. In these cases, restoration costs will provide a gross underestimate of the costs of inaction. [14]

Uncertainty can also complicate efforts to value the costs of inaction. Uncertainty can relate to the ecosystem which is to be valued. For instance, there may be uncertainty about the effect that a specific pressure (*e.g.* concentrations of CO_2) has on negative environmental impacts (*e.g.* sea level rise). With respect to health impacts, there may also be uncertainty about the links between a specific environmental pressure (*e.g.* particulate matter) and impacts on human health (*e.g.* respiratory problems). There may also be uncertainty about the estimated economic value of the anticipated impacts, even if the physical magnitudes of these impacts themselves are known with certainty.[15] And finally, the level of uncertainty is likely to be greater the longer the time horizon over which impacts are to be "costed", with factors such as technological and demographic change being difficult to forecast with precision.

There is therefore considerable uncertainty associated with all stages in the "costing" of the impacts of resource depletion and environmental degradation. It is important to reflect this uncertainty (and risk) in the methodological approach that is adopted, and in the way the results of these studies are communicated. In the presence of significant uncertainty, it is important to assess how much this uncertainty affects the

[14] Some measures taken to mitigate environmental impacts are also irreversible. Kolstad (1996) found that irreversibility in capital investment to mitigate global warming resulted in a less stringent policy due to the benefits of learning. Pindyck (2007) reached a similar conclusion.

[15] There is, of course, also uncertainty with respect to the "costs of action". This can have important implications both for the choice of policy instrument (Roberts and Spence, 1976) for the timing of policy interventions (Pindyck, 2007).

range of possible costs. Depending on the degree of risk aversion that is assumed, the estimated costs of inaction may vary widely.[16]

In methodological terms, at the level of the individual study, it is important to undertake sensitivity analyses in which a broad range of values are applied to those parameters for which there is significant economic or scientific uncertainty. More generally, it may be necessary for policy-makers to draw upon the results of a broad range of models and assessments, since model structure and other factors may be even more important determinants of the estimated costs of inaction than different parameter assumptions.

For some impacts, it may not even be possible to assign credible probabilities to different environmental outcomes. There are some types of potential impacts where "we do not even know what we do not know" (Cole, 2007). In such circumstances, there is a strong case for devoting significant resources toward "investigating seriously the nature of the runaway climate disasters in the thick tails (of the distribution) and what might be done realistically about them" (Weitzman, 2007).

Another important complicating factor associated with evaluating the costs of inaction concerns the treatment of the distributional impacts of environmental degradation. Different types of environmental impact can affect individual countries (and individuals within individual countries) very differently. In some cases, one group of individuals may benefit, while others will bear the costs. Determining the social welfare costs of environmental damages based on estimated individual utility functions ultimately raises basic questions about the weights that are used in the aggregation process.

With decreasing marginal utility of consumption, the distribution of impacts will also affect the aggregate estimate of the costs of inaction. Moreover, there may be good ethical and political reasons (*i.e.* social aversion to inequality) to weight impacts relatively more heavily if they affect poorer households the most.[17] These issues may be particularly relevant in the context of *climate change*. However, social concerns may also relate to specific communities above and beyond distributional implications in terms of income levels. In the area of *natural resource management,* specific concerns of this kind are common (*i.e.* employment in fishing communities).

Summary

There are various possible ways to think about the "costs of inaction". The precise definition to be applied depends on the purpose of the particular study. In turn, the choice of the particular definitions of both "inaction" and "costs" will partly determine the policy use to which the estimates of any particular study can be put. Estimates of the costs of inaction also raise a number of normative issues (including those associated with distributional impacts within and across countries) and analytical issues (discounting, treatment of uncertainty, *etc.*). In the context of the various case studies which follow, these issues are addressed, where they are particularly pertinent.

Cost estimates for some environmental problems will tend to be more readily available in an aggregated sense; while for others, cost estimates may be more readily available for only subsets of costs. For example, estimates of the overall impacts of climate change (*i.e.* the social cost of carbon) will be more

16 Risk and uncertainty are closely associated, but are not identical. "Risk" generally refers to cases in which it is possible to posit probabilities of different outcomes, while "uncertainty" can also reflect cases in which even the set of possible outcomes is unknown.

17 An exposition of the issues involved can be found in Boadway (1976). See also Serret and Johnstone (2006) for a more general discussion of some of the policy implications.

readily available than estimates of the specific costs of climate change with respect to local flooding problems, even though (in principle) the latter is implicit in the former.[18]

From the perspective of a policy-maker concerned with the introduction of new environmental policies, the most appropriate approach will be to think about the marginal social costs and benefits associated with an incremental change in environmental quality, relative to the *status quo* situation (*i.e.* the "counterfactual baseline"). This approach will provide information that can be directly used in decisions about the allocation of scarce resources. However, estimates of the total gross costs of inaction (*i.e.* not the marginal social costs) still have significant value in terms of helping to highlight the economic impacts of not addressing pressing environmental problems. For practical reasons, most of the information provided in the remainder of this report is of the former variety.

18 For instance, it may be methodologically inappropriate to try to disentangle some costs from others.

CHAPTER 2. COSTS OF INACTION WITH RESPECT TO AIR AND WATER POLLUTION

Introduction

Assessing the costs of policy inaction in the area of air and water pollution[19] is complicated by at least four factors:

- The long history of environmental policy in this area, and thus the difficulty associated with defining politically-meaningful baselines against which "inaction" could be measured;

- The local public "bad" aspects of air and water pollution, and thus, the extent to which the costs of environmental degradation are borne privately and publicly;

- The interdependence (both ecological and technological) between different pollutants, and thus, the problem of how to treat ancillary impacts; and

- The heterogeneous nature of costs of inaction in the area of air and water pollution, and thus, some difficulties involved with the aggregation of different impacts.

Air and water pollution were the initial focus of many environmental policies introduced by OECD countries in the 1970s. These were motivated by a perception that natural environments were being degraded at an accelerating rate, with adverse consequences for ecosystems and human health. Measures such as the Clean Air Act (1970) and Clean Water Act (1972) in the US, as well as measures passed by the "Pollution Diet" of Japan in 1970, reflected this rising concern across the OECD. In many cases, the relevant laws passed in this period brought together pre-existing statutes.

Given this long history of environmental policy action with respect to local environmental concerns, evaluating the costs of inaction in the health field is very problematic. Defining what would have been the costs (in terms of degraded air and water quality), in the absence of any policy interventions whatsoever may be, in practical policy terms, meaningless. While there may be some cases in which local environmental concerns have emerged suddenly in the air and water context (resulting in discrete decision points), these are relatively rare.

A distinction must also be drawn between the social and the private costs of inaction. In many cases involving local air and water pollutants, changes in the characteristics of "public" environmental goods are reflected in markets for "private" goods. For example, high concentrations of air pollution or degraded water courses also have implications for other markets – *e.g.* real estate or employment markets. Even in the absence of formal policy "action", there can therefore be (incomplete and imperfect) incentives to take environmental degradation into account in decision-making. As such, some of the costs of inaction will be

[19] This sections draws on previous work undertaken in the context of the project on costs of inaction, including Navrud (2005), and Hepburn (2007), and particularly Scapecchi (2007), Gagnon (2007a) and Gagnon (2007b).

reflected directly in market prices (*e.g.* lower property prices). This can complicate the assessment of the costs of inaction, because of the potential for double-counting.

It can also be difficult to disentangle the costs associated with emissions of a particular pollutant from the costs of emissions of other pollutants. On the one hand, different pollutants may be interdependent (or synergistic) in *ecological* or *epidemiological* terms. For instance, ozone formation is driven by two major precursors: nitrogen oxides (NO_X) and volatile organic compounds (VOCs). When combined in the presence of sunlight, ozone (photochemical smog) is generated. In the absence of one (or the other) pollutant in sufficient concentrations, smog will therefore *not* arise.

On the other hand, air and water pollutants may be interdependent in *abatement* terms – whether as complements or as substitutes. For example, combining flue-gas desulphurization (to reduce SO_2) with selective catalytic reduction (for NO_x) has incidental benefits in terms of mercury reduction. Efforts to reduce one pollutant may therefore result in *increased* emissions of another pollutant. For instance, some measures applied to reduce SO_X, NO_x, and PM have the effect of increasing CO_2 emissions; the ancillary effects in this case will be negative. The costs of "inaction" with respect to each individual pollutant will therefore depend on the level of "action" with respect to the other pollutant.

The costs of inaction in the area of air and water pollution are heterogeneous, and include a wide variety of "use" and "non-use" values. Environmental degradation affects both ecosystem health and human health. Through its impacts on ecosystems, the costs can be related to use values (*e.g.* the effects of ambient ozone on agricultural productivity) or non-use values (*e.g.* the existence value of affected species habitats). The costs can be further distinguished between costs which are reflected in existing market "prices" for different goods and services (*e.g.* lost employee productivity, medical costs, increased raw water treatment costs) and those which are not (*e.g.* health costs in terms of pain and suffering). Table 1 provides a list of selected costs from policy inaction in the areas of air and water pollution.

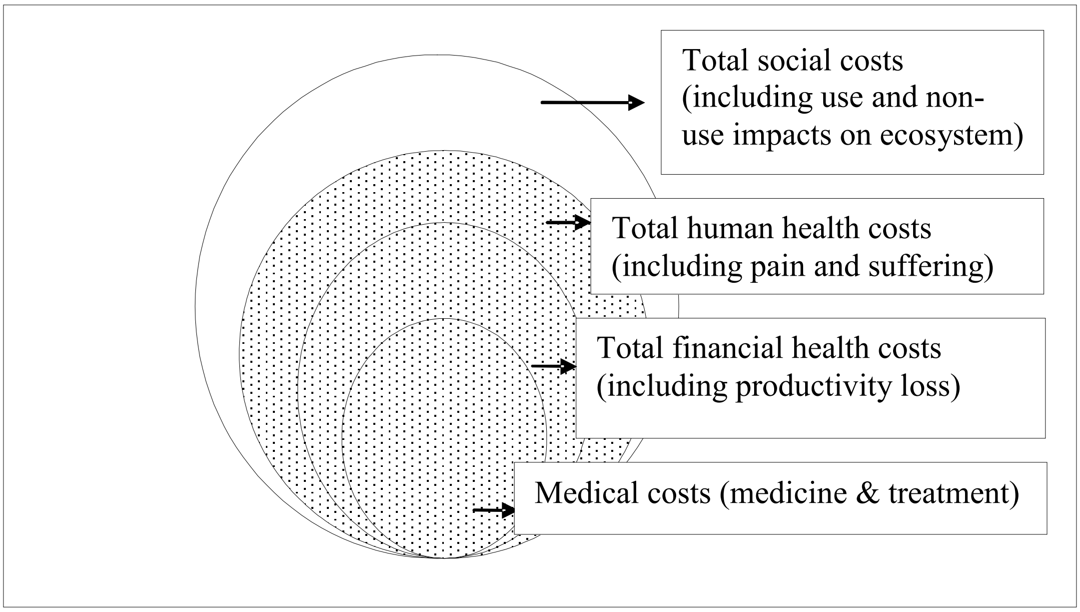
Table 1. Selected Types of Costs Related to Air and Water Pollution

Air Pollution	Water Pollution
Material damages (including heritage)	Increased drinking water treatment
Reduced agricultural yields	Reduced commercial fish stocks
Polluted freshwater sources	Reduced recreational opportunities
Reduced visibility	Loss of biodiversity
Loss of biodiversity	Adverse health impacts
Adverse health impacts	

Table 1 also illustrates that evaluating the total costs of policy inaction with respect to local environmental degradation necessitates aggregating costs which are heterogeneous – in terms of the nature of the underlying impact, the form which such costs take in the economy, and the make-up of those who actually bear the costs. Indeed, the simple summation of relevant values is methodologically inappropriate – because many apparent costs are simply transfers of resources (*i.e.* pecuniary costs), and others embed other (non-environmental) costs within them. While all impacts from policy inaction in the area of water and air pollution are potentially difficult to value, unquestionably the most difficult are impacts on *ecosystems* (*e.g.* airsheds, water courses), which are not directly related to some downstream economic activity.

Although there are a wide variety of impacts from air and water pollution, human health costs often dominate the total costs of environment-related air and water pollution. This means that *lower-bound* estimates of the costs of inaction can be derived on the basis of the valuation of the human health costs of inaction (the shaded areas in Figure 3).

Figure 3. Costs of Inaction with Respect to Air and Water Pollution



One means of valuing health costs is to calculate the costs of medical services and medicines (publicly or privately provided) associated with some environmental pressure. This is known as a "cost of illness" approach, and captures those costs reflected in the innermost circle in Figure 3. In some cases, "cost of illness" studies also seek to value losses in productivity associated with illness, which can be considerable. In the most comprehensive "cost of illness" studies, the loss of productivity of family and other caregivers is also included. Such costs are reflected in the second circle in Figure 3. Many of the studies cited below adopt such a methodology.

Many of the more intangible health costs of environmental degradation are difficult to value, and will not be reflected in any market. For instance, the "personal pain and suffering" associated with being ill will not generally be fully reflected in financial expenditures. Where these costs are significant -- and the empirical evidence suggests that they frequently are -- it is important to rely on stated preference techniques. Health economists generally estimate health impacts in terms of "Quality-Adjusted Life Years" (QALYs), which are derived from various indices of "health states", while environmental economists favour estimates of WTP or WTA for a given change in the risk of experiencing a particular health outcome, expressed in monetary terms (Hammitt, 2007).

Given that health costs can be a significant proportion of the total costs of inaction with respect to air and water pollution, environmental policy in this area can be understood as form of "upstream prevention". The costs of inaction with respect to not undertaking prevention *ex ante* are then reflected in the health costs that are borne *ex post*. However, the *incidence* of the costs associated with these health impacts varies. Table 2 provides an indication of the incidence of types of health impact. Working down the rows, there is a shift toward the innermost circles in Figure 3. Note, however, that the costs are not strictly additive, and it would be inappropriate to sum different costs derived from different studies, in order to estimate "total" health costs.

Table 2. Types and Incidence of Health Costs from Air and Water Pollution

Cost	Examples	Incidence
Pain and suffering	Direct welfare loss	Individual sufferer
Restricted activity	Inability to undertake certain physical activities	Individual sufferer, dependents
Lost productivity	Sick leave, less efficiency	Individual sufferer, employer, insurance (public and/or private)
Preventive behaviour	Residential location, bottled water, lead-free paint	Individual sufferer
Caregiver resources	Compassionate leave, time and effort	Family/friends, employer
Medical service costs	Admission costs, operating costs	Individual sufferer, health insurance, public health service costs
Medicines	Prescription costs	Individual sufferer, health insurance, public health service costs

Empirical estimates of the health costs of inaction, relative to the total costs of inaction, can be documented through a comparison of different air pollution valuation studies. Table 3 indicates that estimated health costs are typically more than 80% of total costs, and sometimes much more. However, only a sub-set of non-health costs are included in the studies in which the health costs exceed 90%. For instance, in the Dziegielewska and Mendelsohn (2005) study, ecosystem and cultural heritage costs comprise more than 13% of total damage, and these costs are not included at all in the other studies. Bearing these general caveats in mind, the focus of this Chapter is on health costs, with some of the other costs being discussed only briefly in the final Section.

Table 3. The Relative Importance of Health Costs in Total Social Costs of Policy Inaction

Study	Context	Non-health costs included	Health %
ECOTECH (2001)	Gothenburg Protocol for Europe (SO_2, NOx, NH_3, NMVOC, PM_{10}, CO, CO_2)	Materials, ecosystem	89%
Dziegielewska and Mendelsohn (2005)	25% improvement in air quality in Poland	Visibility, materials, cultural heritage, ecosystems	82%
USEPA (1999)	Benefits of Clean Air Act in US (NOx, VOC, SO_2, PM_{10}, $PM_{2.5}$, CO)	Materials, visibility	96%
AEA (1999)	Gothenburg Protocol (SO_2, NOx, VOC, NH_3) in Europe	Materials, crops, timber	95%
Olsthoorn *et al.* (1999)	SO_2 - 50,000 tonnes (10%) reduction in Netherlands	Materials	97%
Muller and Mendelsohn (2007)	PM, NOx, NH_3, SO_2, VOC in the United States	Agriculture, Visibility, Materials, Recreation	94%

The health implications of air and water pollution are extremely varied, and therefore do not lend themselves to brief summary. However, in the following Sections, some of the main concerns are reviewed, first for water pollution and then for air pollution.

Water pollution and health

Water pollutants can be disaggregated into three broad groups: disease-causing bacterial pollutants; oxygen-demanding pollutants; and water-soluble inorganic pollutants. The first and third of these have significant health implications. The main sources include municipal wastewater collection and treatment systems, runoff from agricultural practices, and effluent from manufacturing facilities. Particular industrial sectors in which the potential contribution to water pollution is significant include the chemicals sector, the food and beverage sector, and the pulp and paper sector. In addition, the mining and mineral processing sectors can have significant implications for water quality, as can direct household discharge of hazardous substances into drains.

Many of the costs of inaction related to water pollution exhibit health (not just environmental) externalities. For example, the likelihood of an individual experiencing a case of diarrhoea is affected by the prevalence of diarrhoea in the household or local community more generally. There are therefore two externalities for infectious waterborne diseases – the environmental externality (which relates to levels of exposure) and the public health externality (which relates to disease transmission). This is different from the health impacts of air pollution, which only embody the former.

Significant strides have been made in recent years in terms of addressing water pollution. The policy framework for the regulation of industrial point sources is well-developed in most OECD countries, although some pollutants, such as heavy metals and chlorinated solvents, remain a concern. Increasing attention is being paid to "non-point" sources such as agricultural runoff, which are more difficult to regulate. High levels of nutrients such as nitrogen and phosphorous in water can cause rapid growth of phytoplankton, creating dense populations, or blooms. These nutrients occur naturally in soil, animal waste, and plant material, and as such agricultural run-off is an important contributor.

In addition to efforts to reduce run-off of organic pollutants from fertilisers and manure, organophosphates and carbonates from pesticides are of concern. While the percentage of the population connected to sewerage systems and the level of sewage treatment has increased in OECD countries in recent decades, there are still deficiencies in collection and treatment systems in some countries (OECD, 2005a). Total investment in the water sector for the 30 OECD countries, which already exceeds USD 150 billion per year (over 0.5% of GDP), is likely to increase further in the years ahead (OECD, 2001).

Low quality of raw water for drinking water is an important source of adverse health impacts. The quality of raw water supply is closely related to the quality of sewage treatment, including urban storm water runoff management. For example, in the US, "drinking water outbreaks have been linked to runoff; more than half of the documented waterborne disease outbreaks between 1948 and 1994 followed extreme rainfalls" (Curriero *et al.,* 2001; Garfield *et al.,* 2003). Thus, better treatment of urban storm water and wet-weather overflows of sewage would reduce health impacts considerably (OECD, 2003). Agricultural runoff is another important source of organic pollutants. Changes in agricultural production techniques, including high-density animal operations carried out in proximity to urban areas, have led to an increase in the transmission of animal pathogens to humans (Payment and Hunter, 2001).

In some cases, disinfectants used in water treatment have been inadequate in the control of some of the most common waterborne pathogens. The simple use of chlorine disinfection may not suffice for the elimination of most bacterial waterborne pathogens, due to the parasites' increased resistance (Payment and Hunter, 2001). Resistance to antibiotics by waterborne disease therefore represents a major public health threat in OECD countries (Levin *et al.*, 2002; Payment and Hunter, 2001). Lower immunity to pathogens may also be a consequence of improved sanitary conditions (Payment and Hunter, 2001). This is exacerbated by the increase in the number of susceptible individuals, especially the elderly (Levin *et al.*, 2002).

At the global level, diarrhoeal diseases are estimated to be the largest contributors to the burden of water-related disease. Infectious diarrhoea can be caused by bacteria (*e.g.* cholera, E. coli, shigellosis, typhoid fever), viruses (*e.g.* norovirus, rotavirus), protozoan parasites (*e.g.* amoebiasis, cryptosporidiosis, giardiasis). The greatest risk from pathogen micro-organisms (*i.e.* bacteria, viruses, parasites and helminths) is associated with consumption of drinking water in both developing and developed countries (WHO, 2004a). Inadequate treatment or disinfection of drinking water can result in contamination, subsequently affecting human health. OECD countries are subject to large-scale contamination of waterborne disease when water supply safety is compromised (*e.g.* by a breakdown in treatment systems), which may lead to detectable disease outbreaks. Recurring contamination may lead to sporadic diseases, which public health surveillance systems may not correctly attribute to drinking water (WHO, 2004a).

Human faecal pollution can also affect the quality of recreational waters, leading to health problems (WHO, 2003). Numerous epidemiological studies have shown that exposure to recreational waters contaminated with faeces can result in several types of illness, including gastroenteritis, acute respiratory disease, and infections of eyes, ears and skin (Prüss-Ustün, 1998; Dwight *et al.*, 2005; WHO, 2003). Microbial contamination of recreational water with sewage is widespread and affects a large number of users worldwide. Moreover, "direct discharge of crude, untreated sewage (for instance, through short outfalls or combined sewer overflows, which contain a mixture of raw sewage and storm water) into recreational areas present a serious risk to public health" (WHO, 2003).

Finally, chemical contamination of source water is an increasingly important issue, compromising conformity with mandatory health standards for drinking water (OECD, 2006a). The effects of chemical contamination tend to be chronic (WHO/EURO, 2004). High nitrate, fluoride, or arsenic concentrations can have significant health impacts, namely methaemoglobinaemia (blue baby syndrome), dental fluorosis, and skin lesions, respectively. Several European OECD countries have reported high nitrate concentrations in drinking water (WHO/ECEH, 2005; EEA-WHO/EURO, 2002). In addition, other chemical pollutants are of concern in this region, such as chloroform, fluoride, arsenic, trihalomethanes, pesticides, boron, copper, lead, nickel, tetrachoroethene and trichloroethene (WHO/ECEH, 2005) (Table 4).

Table 4. Health Effects Associated With Selected Water Pollutants

	Disease/Pollutant	Health impacts
Bacterial	Amoebic dysentery	Abdominal pain, diarrhoea, dysentery
	Capbylobacteriosis	Acute diarrhoea
	Cholera	Sudden diarrhoea, vomiting. Can be fatal if untreated
	Cryptosporidiosis	Stomach cramps, nausea, dehydration, headaches. Can be fatal for vulnerable populations.
Chemical	Lead	Impairs development of nervous system in children; adverse effects on gestational age and fetal weight; blood pressure
	Arsenic	Carcinogenic (skin and internal cancers)
	Nitrates and nitrites	Methaemoglobinaemia (blue baby syndrome)
	Mercury	Mercury and cyclodienes are known to induce higher incidences of kidney damage, some irreversible
	Persistent organic pollutants	These chemicals can accumulate in fish and cause serious damage to human health. Where pesticides are used on a large-scale, groundwater gets contaminated and this leads to the chemical contamination of drinking water.

Source: EEA/WHO-Europe (2002).

Air pollution and health

In the case of air pollution, high concentrations of particulate matter (PM), carbon monoxide (CO), nitrogen dioxide (NO_2), sulphur dioxide (SO_2), volatile organic compounds (VOC) and ozone (O_3) all have adverse implications for human health, although in most cases the epidemiological evidence is uncertain and further research efforts are underway to better understand the links. (Table 5 lists some of the impacts for which there is good evidence.) Air pollution is caused by both natural and anthropogenic sources. The anthropogenic sources of pollutants in ambient air can be either mobile or fixed.

Significant anthropogenic sources of ambient air pollution include industries, transport, and power generation[20]. The most common source of air pollution is the burning of fossil fuels in power stations, industries, buildings and houses, and road traffic. Fossil fuel combustion is responsible for emissions of NO_2, SO_2, CO, PM, VOC and lead as well. Other sources include wildfires, chemical products, fertiliser and paper production as well as waste incineration. In Europe, the greatest contributors to emissions of primary PM_{10} and gases leading to the formation of secondary PM_{10} in 2000 were the energy-production (30%), road-transport (22%), industrial (17%) and agricultural (12%) sectors (Krzyzanowski *et al.*, 2005).

In addition, and as noted above, O_3 is the result of a photochemical reaction of sunlight on VOCs, in the presence of NO_2. As such, O_3 is referred to as a "secondary" pollutant. There are also indirect sources of PM emissions, created by the combination with other gases such as NO_x (nitrates) and SO_x (sulphates). Therefore, PM pollution can be considered as either a primary or a secondary pollutant.

Table 5. Health Effects Associated With Selected Air Pollutants"

Pollutant	Short-term effects	Long-term effects
PM	- Increase in mortality - Increase in hospital admissions - Exacerbation of symptoms and increased use of therapy in asthma - Cardiovascular effects - Lung inflammatory reactions	- Increase in lower respiratory symptoms - Reduction in lung function in children and adults - Increase in chronic obstructive pulmonary disease - Increase in cardiopulmonary mortality and lung cancer - Diabetes effects - Increased risk for myocardial infarction - Endothelial and vascular dysfunction - Development of atherosclerosis
O_3	- Increase in mortality - Increase in hospital admissions - Effects on pulmonary function - Lung inflammatory reactions - Respiratory symptoms - Cardiovascular system effects	- Reduced lung function - Development of atherosclerosis - Development of asthma - Reduction in life expectancy
NO_2	- Effects on pulmonary structure and function (asthmatics) - Increase in allergic inflammatory reactions - Increase in hospital admissions - Increase in mortality	- Reduction in lung function - Increased probability of respiratory symptoms - Reproductive effects

Source: Adapted from WHO (2004b; 2006).

Emission intensities for different pollutants show significant variation across OECD countries, depending mainly on national economic structure and energy consumption patterns (OECD, 2005b). Compared to 1990 levels, SO_x have decreased significantly in all but a few countries. European countries have in general achieved more significant reductions in SO_x emissions because of earlier commitments. The Gothenburg Protocol (adopted in both Europe and North America) should further reduce SO_x emissions in the years ahead.

Reductions of NO_x emissions have been less important (and have actually risen in recent years), suggesting only a weak decoupling from GDP compared to 1990 (OECD, 2004). Important variations in NO_x

20 In the European Union, road transport and energy industry contribute to about 27% of the total emissions of PM10 (Krzyzanowski *et al.*, 2005).

emission intensities over time can be observed among OECD countries. Emission reductions were particularly significant in the early 1990s in many European countries, because of the Sofia Protocol, designed to stabilise NO_x emissions by the end of 1994 to their 1987 levels. However, some European OECD countries have not yet met these objectives, and the achievement of further reductions (as described in the Gothenburg Protocol) will require additional efforts.

CO levels in ambient air have decreased, mostly as the result of the introduction of new standards and equipment in transport and manufacturing. Examples include the introduction of catalytic converters for cars, and stricter standards for fuel quality specifications for petrol and diesel fuels (EURO IV and V). These policies have also implied a significant decrease in VOC emissions. However, additional measures will have to be undertaken to meet the objectives of the Gothenburg Protocol (to reduce VOC emissions by 56% in 2010 in relation to 1990 emission levels).

PM_{10} emissions have also significantly decreased. Emissions of PM_{10} are expected to be further reduced in the years ahead, as improved vehicle engine technologies are adopted (Euro V) and stationary fuel combustion emissions are controlled through the abatement or use of low-sulphur fuels, such as natural gas.

Human exposure to these pollutants is particularly high in urban areas where economic activities and road traffic are concentrated. Of growing concern are the concentrations of fine particulates, NO_2, toxic air pollutants, and acute ground-level ozone pollution episodes in both urban and rural areas. These pollutants can interact. For instance, it has been estimated that the proportion of lung cancer attributable to toxic and carcinogenic air pollutants inhaled via particulate matter may be as high as 10% in Europe (Boffetta, 2006).

Table 6 presents concentrations of selected air pollutants for OECD countries in 2002. Average urban concentrations were estimated in residential areas of cities larger than 100,000 (World Bank, 2006a); some of the OECD cities are reported here. These concentration levels can be compared with WHO guidelines on air quality (WHO, 2005), which recommend the following ranges of values: $PM_{2.5}$: 10 $\mu g/m^3$ annual mean; PM_{10}: 20 $\mu g/m^3$ annual mean; O_3: 100 $\mu g/m^3$ for daily maximum 8-hour mean; NO_2: 40 $\mu g/m^3$ annual mean; and SO_2: 20 $\mu g/m^3$ for the 24-hour mean.

Based on projections from the OECD *Environmental Outlook* (2008), the potential exposure of the urban populations to ozone concentrations is presented in Figure 4. Following the recommendations of the WHO, the ozone levels are expressed as ozone concentrations above 35 parts per billion. An increase in ozone exposure is seen globally. At a global level, a 25% increase is expected to 2030, but this varies between regions from less than a 5% increase, to more than 55%.

Table 6. Air Pollution Concentrations in PM_{10}, SO_2[21] and NO_2, for 2002

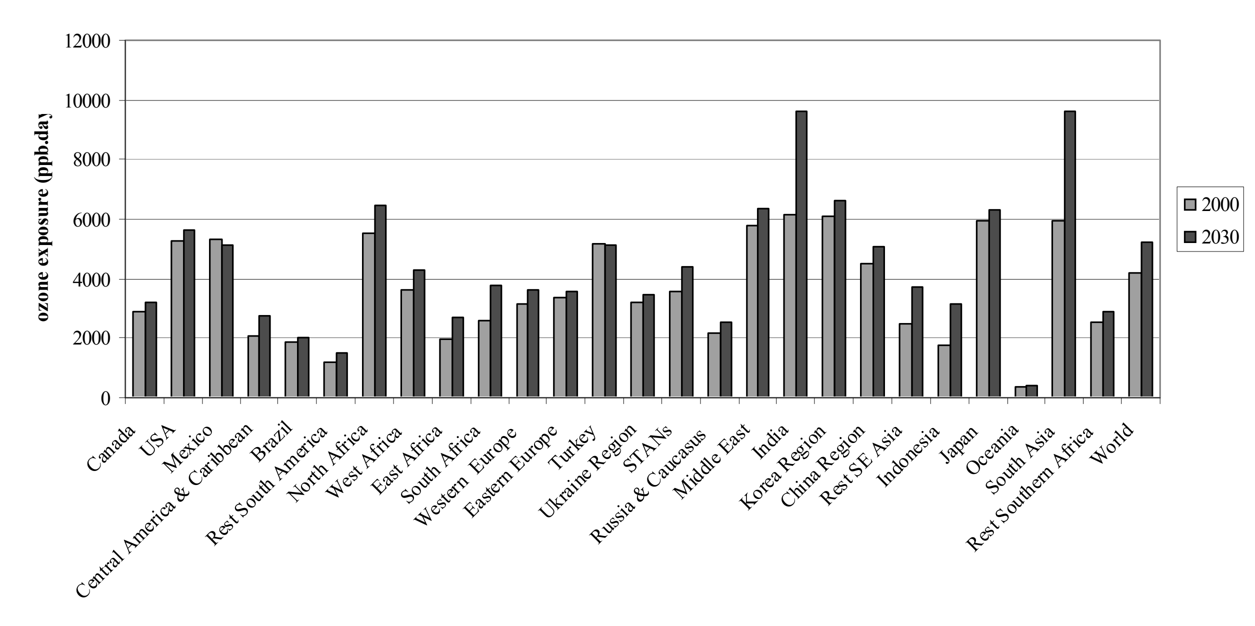
Countries	City	Average annual concentration of PM_{10}, μg/m³	Average annual concentration of SO_2, μg/m³	Average annual concentration of NO_2, μg/m³
Australia	Melbourne	13		30
	Perth	13	5	19
	Sydney	**22**	**28**	**81**
Austria	Vienna	**44**	14	**42**
Belgium	Brussels	**30**	20	**48**
Canada	Montreal	**20**	10	**42**
	Toronto	**24**	17	**43**
	Vancouver	14	14	37
Czech Republic	Prague	**25**	14	33
Denmark	Copenhagen	**23**	7	**54**
Finland	Helsinki	**23**	4	35
France	Paris	12	14	**57**
Germany	Berlin	**25**	18	26
	Frankfurt	**22**	11	**45**
	Munich	**22**	8	**53**
Greece	Athens	**51**	**34**	**64**
Hungary	Budapest	**23**	**39**	**51**
Iceland	Reykjavik	20	5	**42**
Ireland	Dublin	**21**	**20**	
Italy	Milan	**36**	**31**	
	Rome	**35**		
	Torino	**53**		
Japan	Osaka	**37**	19	**63**
	Tokyo	**42**	18	**68**
	Yokohama	**32**	**100**	13
Korea	Pusan	**44**	**60**	**51**
	Seoul	**46**	**44**	**60**
	Taegu	**50**	**81**	**62**
Mexico	Mexico City	**55**	**74**	**130**
Netherlands	Amsterdam	**40**	10	**58**
New Zealand	Auckland	15	3	20
Norway	Oslo	19	8	**43**
Poland	Lodz	**39**	**21**	**43**
	Warsaw	**43**	16	32
Portugal	Lisbon	**28**	8	**52**
Slovakia	Bratislava	19	**21**	27
Spain	Barcelona	**43**	11	**43**
	Madrid	**37**	**24**	**66**
Sweden	Stockholm	13	3	20
Switzerland	Zurich	**26**	11	39
Turkey	Ankara	**54**	**55**	**46**
	Istanbul	**64**	**120**	
UK	Birmingham	**26**	9	**45**
	London	**23**	**25**	**77**
	Manchester	17	**26**	**49**
US	Chicago	**26**	14	**57**
	Los Angeles	**36**	9	**74**
	New York	**22**	**26**	**79**

Note: The data was obtained from a variety of different sources based on annual average concentrations at monitoring stations. The values reported may differ from those reported in national government statistics publications. For instance, in the case of Spain reported values for Barcelona and Madrid are 47 and 34 for particulate matter, 4 and 10 for SO_2, and 54 and 57 for NO_2.

Source: World Bank, 2006b. WHO (2006) provides more recent data, but only in graphical form.

21 This cannot be directly compared with the WHO annual mean.

Figure 4. Potential Exposure of Urban Population to Ozone Concentrations of More than 35 Parts per Billion



Source: OECD *Environmental Outlook* (2008).

Aggregate health effects from air and water pollution

Assessing the overall health effects of air and water pollution cannot be undertaken with precision. However, in a "rough-and-ready" manner, the WHO has sought to link "environmental" factors with both mortality and disability adjusted life years (DALYs) (Prüss-Ustün and Corvalán, 2006). Table 7 presents some of the main findings. In OECD countries, the contribution of "environmental" mortality to total mortality is just under 20%; for developing countries, it is marginally higher. However, it must be emphasised that the WHO uses a very broad definition of "environmental risk factors", including many factors which are not affected by environmental policy interventions.

Table 7 also presents data on deaths from lower respiratory infections (for which indoor and outdoor air pollution are important contributors), and on diarrhoeal diseases (for which unsafe water supply, sanitation and hygiene are significant contributors). WHO estimates that environmental factors are responsible at the global level for over 40% of mortality from the former, and 90% for the latter. It is striking to note that there are over 3 million deaths per year related to these two causes.

Table 7. Global Burden of Disease from Selected Environmental Risk Factors

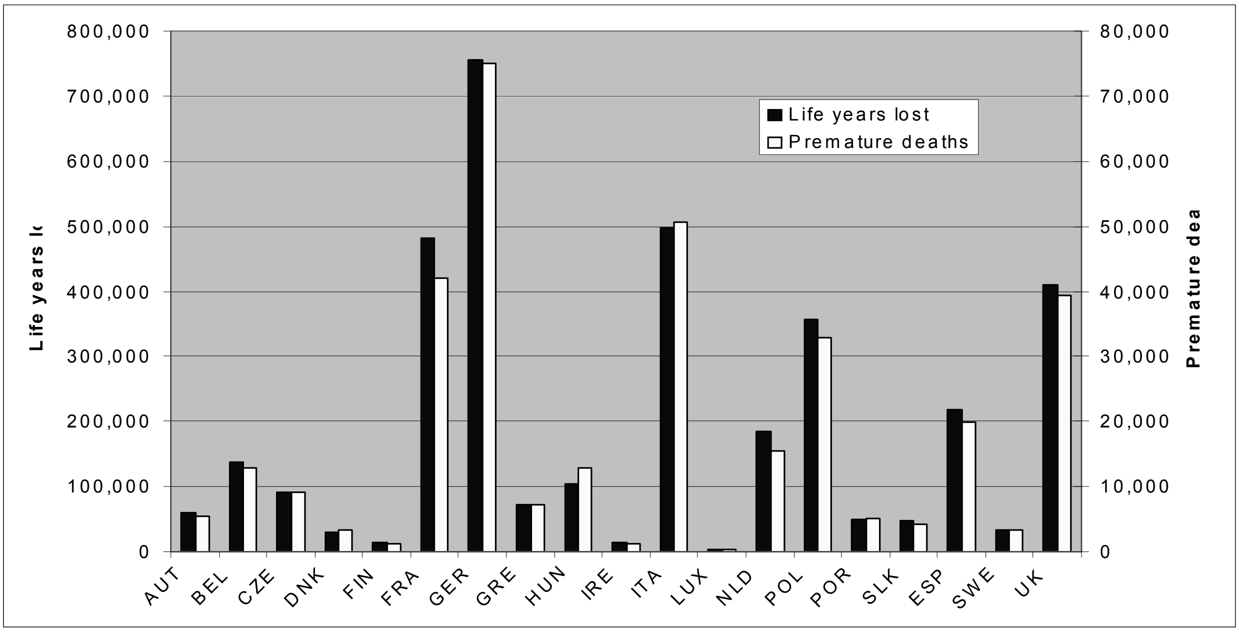
	Developed Countries	Developing Countries
Population ('000s)	1366867	4858118
Total Deaths ('000s)	13430	43599
"Environmental" Deaths ('000s)	2302	10994
Lower Respiratory ('000s)	113	1403
Diarrhoeal ('000s)	18	1664

Source: Prüss-Ustün and Corvalán (2006).

At the global level, outdoor PM pollution has been estimated to be responsible for approximately 800,000 premature deaths (*i.e.* 1.4% of global deaths) and 6.4 million years of life lost (*i.e.* 0.7% of total years of life lost) each year (Cohen *et al.*, 2004). The burden of disease attributable to outdoor air pollution is greatest in developing countries, with 39% of total years of life lost occurring in south-east Asia (*e.g.* China, Malaysia, Cambodia, Viet Nam) and 20% in Asia (*e.g.* India, Bangladesh, Bhutan, Nepal). If both mortality and morbidity aspects are considered, Asian and Eastern European countries (*e.g.* Turkey, Poland, Romania, Slovakia) are the most significantly affected, because urban air pollution is thought to be responsible for 0.7% to 1% of the total burden of disease in these regions (Cohen *et al.*, 2004).

Premature deaths associated with PM_{10} pollution observed in 2002 are presented, in Figure 5, together with DALYs (limited to YLL) for selected European OECD countries. There is wide variation across countries. France, Germany, Italy, Poland and the UK report a relatively high number of premature deaths attributable to PM_{10} pollution (between 40,000 and 75,000 annual deaths), which, together with years of life lost, makes this a major environmental health issue.

Figure 5. Deaths and Years of Life Lost Associated with PM_{10} Pollution in Selected OECD Countries in 2002

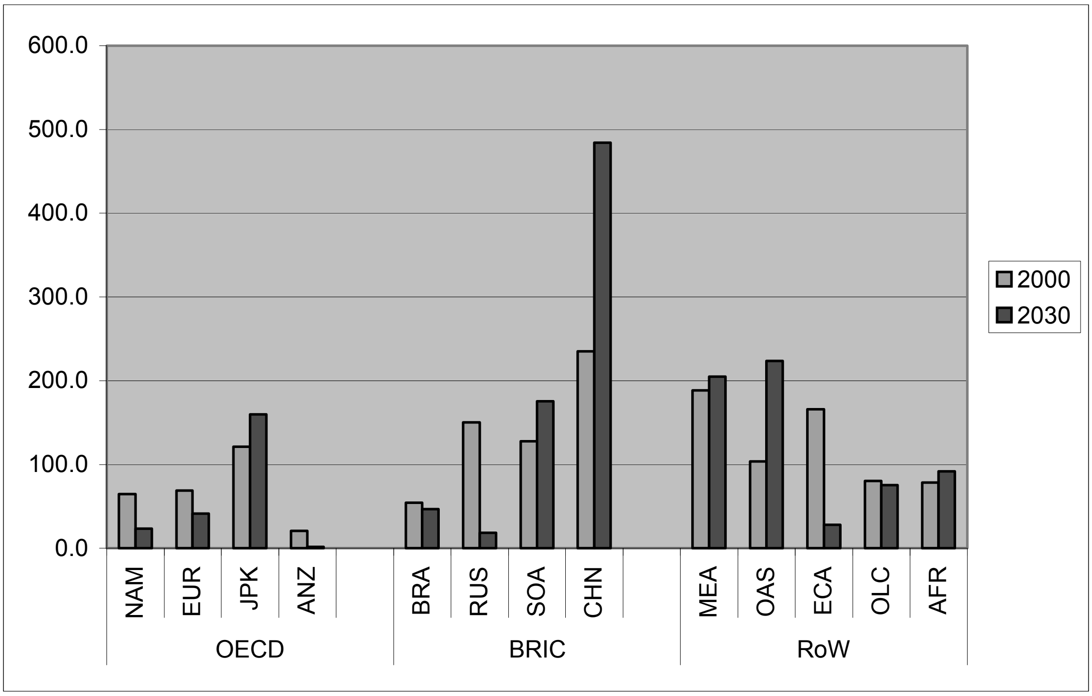


Source: AEA Technology Environment (2005).

Levy *et al.* (2007) have estimated that there is a 0.4% increase in short-term mortality for each 10 ppb increase in 1-hour maximum ozone over the year. In a panel data study covering 23 European countries from 1987 to 2002 Hwang (2007) estimated that a 1% reduction in lagged average concentrations of ozone would have saved 112 infant lives.

According to estimates from the OECD *Environmental Outlook* (2008), the situation is likely to deteriorate at a global level by 2030 – with a 44% increase in premature deaths per million inhabitants in urban agglomerations. In the OECD, there are expected to be 33% fewer premature deaths, while for the BRICs and the Rest of the World, the situation generally deteriorates (Figure 6).

Figure 6. Premature Death per Million Inhabitants in Urban Agglomerations Attributable to Urban Outdoor Exposure to PM10 (Deaths per Million)

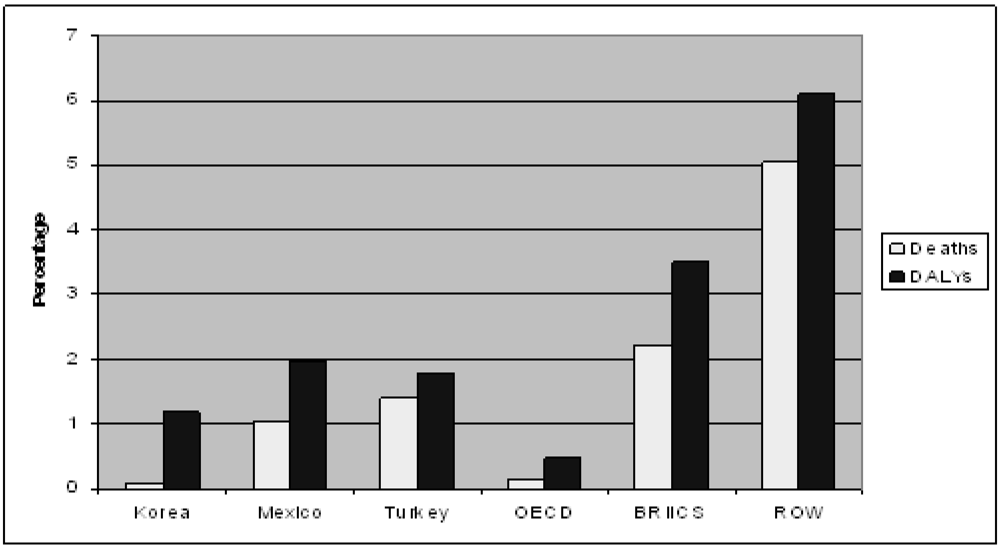


Source: OECD *Environmental Outlook* (2008).

Deficiencies in water supply and sanitation lead to even more significant health impacts. At the global level, about 1.1 billion people do not have access to safe water supply and 2.6 billion people do not have access to adequate sanitation facilities, mainly in developing countries (WHO/UNICEF, 2006). The associated health impacts are alarming: 1.7 million deaths, of which 90% of were children under 5 years old. Indeed, unsafe WSH is the world's biggest child killer after malnutrition. An estimated 88% of all diarrhoeal diseases are attributable to unsafe drinking water supply, inadequate sanitation, and poor hygiene. The BoD attributable to this risk factor is 65.2 million DALYs. 3% of all deaths and 4.4% of all DALYs were attributable to unsafe WSH and were caused by diarrhoeal diseases, schistosomiasis, trachoma, and selected intestinal nematode infections (Gagnon, 2007a, 2007b).

The figures for OECD countries are generally lower – but some OECD countries are still significantly affected. Figure 7 gives mortality and DALYs for unsafe WSH in Korea, Mexico and Turkey relative to the OECD, BRIIC and ROW. Three-quarters (76%) of all deaths attributable to diarrhoeal disease in OECD countries in 2002 occurred in Mexico and Turkey, with Korea also being significantly affected in terms of DALYs. Nonetheless, less than 1% of all deaths attributable to diarrhoeal disease at the global level occurred in OECD countries.

Figure 7. % of Total Mortality and Burden of Disease Due to Unsafe Water, Sanitation and Hygiene – 2002



Source: Gagnon (2007a and 2007b).

Valuation of the health costs of policy inaction with respect to air and water pollution

Many empirical studies have sought to value in monetary terms the health benefits of policy interventions (or costs of policy inaction). A summary of some of the key studies related to water and air pollution is presented below.

Many empirical studies have sought to value in monetary terms the health benefits of policy interventions (or costs of policy inaction). A summary of some of the key studies related to water and air pollution is presented below.

Valuation of health impacts of water pollution

Table 8 provides estimates of the health benefits of selected policy interventions related to water pollution in OECD countries. Many of these are *lower-bound* estimates, since they are obtained from cost-of-illness studies which do not account for pain and suffering. In some cases, non-financial opportunity costs for caregivers and other third parties are not included either.

1. A number of these valuation exercises were undertaken as part of cost-benefit analyses (CBA). While this report focuses on the costs of inaction (which is analogous to only the "benefits" side of a CBA), it is nonetheless instructive to compare the ratio of benefits of a policy intervention (costs of inaction) to the costs for the policy scenarios assumed in the various studies (Table 9). In each of the cases reported in Table 9, the benefits of the policy intervention are at least equal to the costs, and in some cases, the benefits are an order of magnitude greater than the costs. This is particularly striking, since these studies only include the "health" benefits. However, not all water policy interventions have benefit/cost ratios in excess of unity. Indeed, Freeman (2002) cites a number of examples in which policies could not be justified on health grounds alone, unless very high assumptions of VSL or morbidity benefits are assumed. The EPA (2001) study cited above provided benefit/cost ratios for more stringent standards for arsenic, with the ratio being less than unity for standards of 5 μg/l. This highlights the importance of a careful balancing of the costs of inaction, relative to the benefits of inaction (avoided compliance costs).

Table 8. Valuation of Health Benefits of Selected Policies Related to Water Pollution

Scenario assessed	Studies	Benefits of Policy Intervention/Costs of Policy Inaction
Health benefits of improving the quality of coastal water in Estoril Coast (Portugal)	Machado and Mourato (1999)	USD 7.1 - 10.7 million per year
Health benefits of quality improvement of recreational waters in south-west Scotland (UK)	Hanley *et al.* (2003)	GBP 1.3 million per year
Health benefits of improving the quality of recreational waters in Brest harbour (France)	Le Goffe (1995)	EUR 33.23 per household per year
Health costs associated with urban runoff in recreational waters in Orange County, California (USA)	Dwight *et al.* (2005)	USD 3.3 million per year
Health costs of the cryptosporidiosis outbreak in Milwaukee (USA)	Corso *et al.* (2003)	USD 96.2 million in 1993
Health benefits of improving drinking water quality in the USA	USEPA (2006a)	USD 130 million – 2.0 billion
Improving the drinking water quality and storm water management in the US	Garfield *et al.* (2003)	USD 2.1-13.8 billion per year
Reducing chemical contamination of drinking water (Korea)	Kwak and Russell (1994)	USD 106 million/year
Improving the quality of recreational waters in the UK	Georgiou *et al.* (2005)	25% reduction of illness: GBP 11.9 billion/ 100% reduction: GBP 22.8 billion for a 25-years period
Improving the quality of recreational waters in the Netherlands	Brouwer and Bronda (2005)	EUR 2.4 billion for a 20-year period
Health benefits associated with reducing arsenic from 50 μg/l to 10 μg/l	US EPA (2001)	USD 139.6 million – USD 197.7 million/year
Reduction of nitrate exposure in the US to legal safety standards	Crutchfield *et al.* (1997)	USD 350 million per year.
Reduction in copper level in drinking water from 4.3/ mg.l to 1.3 mg/l in SW Minnesota	Kim and Cho (2002)	USD 1.66 million – USD 2.38 million

Table 9. Benefit-Cost Ratios for Selected Water-related Studies

	Unfavourable Assumptions	Favourable Assumptions
USEPA (2006a)	1.0:1	30.1:1
Kwak and Russell (1994)	1.0:1	4.8:1
Georgiou *et al.* (2005)	2.3:1	9.5:1
Brouwer and Bronda (2005)	48.0:1	
USEPA (2001)	1.0:1	1.1:1

In a study commissioned by the WHO, the Swiss Tropical Institute (Hutton and Haller, 2004) outlined the significant benefits of improving water supply, sanitation facilities and hygiene behaviour in both developing and developed countries. Estimates of the value of health impacts were based on the estimated BoD attributable to unsafe WSH in 2000 (see Prüss-Üstün *et al.,* 2004). These are COI estimates, and thus the benefits of reduced intangible costs (*e.g.* pain and suffering) are not included. As a result, they clearly represent an underestimation of the health costs of unsafe WSH.

Among the interventions analysed, attainment of the UN Millennium Development Goals for water supply, as well as the World Summit on Sustainable Development (WSSD) goal for sanitation, which are to halve the proportion of people who do not have access to improved water sources and improved sanitation facilities, was assessed. Total benefits of meeting the objective are estimated to be USD 128.92 billion annually (using a minimum wage approach).[22]

The principal contributors to the estimates of total benefits were the savings in time costs involved in securing adequate access to adequate water and sanitation services. These amounted to approximately 89% of the total benefits (avoided costs), using a minimum wage approach for the opportunity cost of time.[23] While a proportion of these benefits may be considered "defensive" health expenditures associated with environmental factors, it would be inappropriate to consider them to be wholly health-related. At the global level, the health benefits of achieving the MDG objectives relative to the existing level of service provision are estimated to be USD 14.338 billion annually. These are decomposed into the different elements in Figure 8.

Conversely, the estimated financial cost of meeting the international commitment to the MDG goals is estimated to be USD 11.305 billion annually. The cost estimates provided by Hutton and Haller (2004) are low compared to other assessments, which assume household connections in urban areas[24]. Of course, the estimates of the benefits would also be considerably higher with even better levels of service provision.

The total (global) estimated economic benefits of meeting international commitments for water and WSSD goals for sanitation are, therefore, higher than the costs of the required water and sanitation improvement. Consequently, every dollar invested in meeting the water MDGs and WSSD for sanitation is estimated to reduce health costs by USD 1.27 at the global level (even without adding in the intangible health impacts related to suffering and pain), and by USD 11.4 in term of total costs. There are clearly significant costs associated with inaction in this policy field.

In all developing countries regions, the total benefits which would arise out of meeting the water MDG goals and the WSSD goal for sanitation outweigh the associated implementation costs. In Africa[25], for instance, the estimated annual cost of achieving these targets is USD 2.02 billion, while the total benefits (avoided costs) were estimated to be USD 21.73 billion in 2000, of which USD 4.74 billion were

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22 These are 2006-adjusted values of the WHO report (Hutton and Haller, 2004), provided directly by Dr. Guy Hutton from the Swiss Tropical Institute. They are based on further improvements to the model (personal communication: 16 March, 2006). The differences lie in the calculation method -- the annual time savings estimates calculated in 2004 using a "GNP per capita" base, whereas the new ones were calculated using a "minimum wage" base. Originally, the total social benefits estimated (in 2004) were USD 84.4 billion (using a GDP per capita approach).

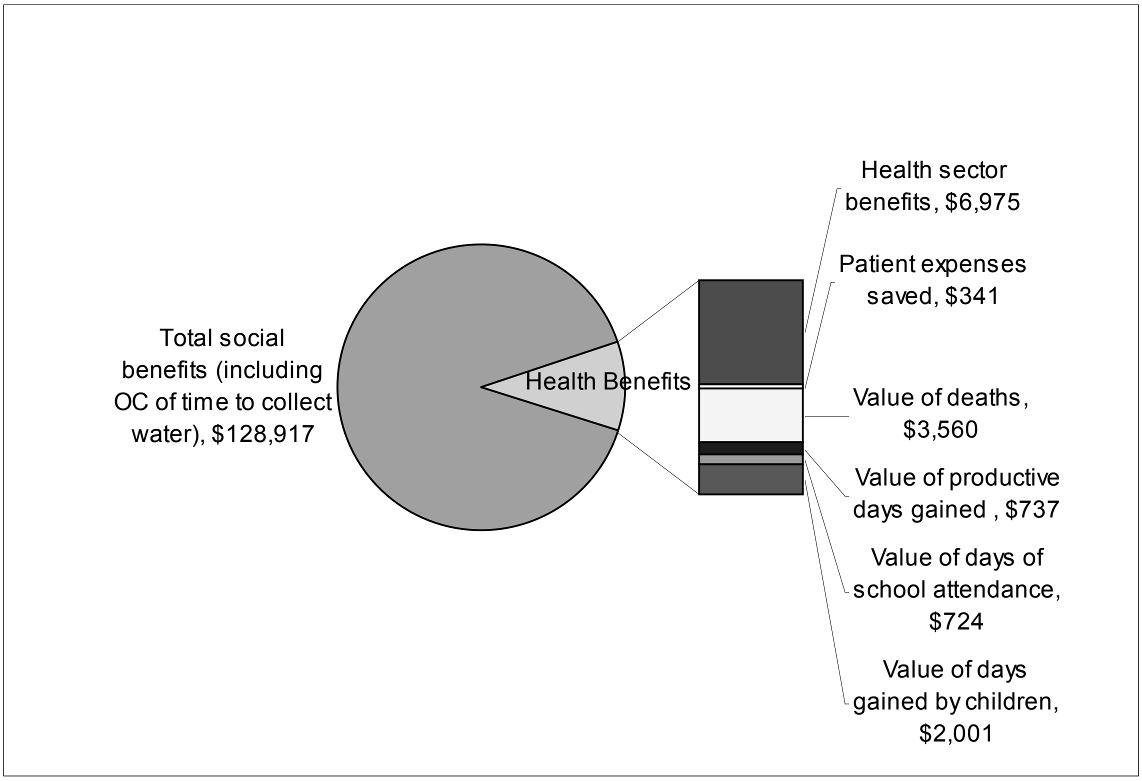
23 Value-of-time savings due to access to water and sanitation: USD 63 547 million (using GDP per capita) or USD 114 579 million (using minimum wage).

24 The cost estimates are comparable to those provided in the Camdessus report (2003), as they consider low-cost technologies. In a review of six global assessments, WWC (2006) demonstrated that the investment cost to achieve the Target 10 MDG on water and sanitation range from 9 billion to 30 billion USD per year. This highlights that the results are only comparable if they are analysed on comparable bases. The Hutton and Haller estimates are lower, since they are based on low-cost technology. For more details, see WWC (2006).

25 This region includes AFR-D and AFR-E, as defined by WHO epidemiological sub-regions. Consequently, Egypt, Morocco and Tunisia are not included.

associated with reduced health costs. Avoided health costs significantly outweigh the implementation cost of the water and sanitation improvement to reach MDG Target 10 for water supply and sanitation in Africa.

Figure 8. Decomposition of Costs of Not Meeting the MDGs for WSS (USD million)



Source: Hutton and Haller (2004).

Table 10 provides estimates of the monetised health benefits (avoided health costs) and the total social benefits (avoided social costs) associated with three other interventions for improving water supply and sanitation facilities at the global level. In this case, the "total benefits" column includes the avoided costs associated including monetised time savings. Implementation costs vary widely, due to the level of "improvement" assumed.

While the estimates for both the numerator and the denominator are controversial, it is interesting to see that the BCR is significantly greater than 1 in all cases when the opportunity costs of time are included. The programme with the highest ratio in terms of health benefits (avoided health costs) is the one which compares the existing situation with the case where water is disinfected at the point of use for all, as well as improved water and sanitation services. Access for all to a regulated piped water supply and household sewage connections is also found to be an efficient intervention at the global level; however, it cannot be justified on health grounds alone.

The ratio of benefits (avoided costs) to implementation costs for water supply and sanitation interventions is particularly high in developing countries. This is due in part to the fact that the avoided health costs from improving services are generally higher when the initial quality of the environment is poor, compared to the case where the environment is of higher quality. Moreover, the interventions proposed in OECD countries to further improve drinking water quality and waste water treatment are more costly than what is needed in developing countries (*e.g.* basic water supply and sanitation facilities). The costs of inaction are therefore particularly high in developing countries.

Table 10. CBA of Improving Water Supply and Sanitation at the Global Level per Year[26]

Environmental interventions	Implementation Costs	Health Benefits (Avoided Health Costs)	BCR (just health benefits)	Total benefits	BCR (total benefits)
Halving the proportion of the population who do not have access to improved water sources and improved sanitation facilities (MDG for water and sanitation)	USD 11.3 billion	USD 14.3 billion	1.3	USD 128.9 billion	11.4
Access for all to improved water and improved sanitation	USD 22.6 billion	USD 23.3 billion	1.0	USD 252.5 billion	11.2
A minimum of water disinfected at the point of use for all, on top of improved water and sanitation services	USD 24.6 billion	USD 77.3 billion	3.1	USD 306.5 billion	12.5
Access for all to a regulated piped water supply and sewage connection into their houses	USD 136.5 billion	USD 100.9 billion	0.7	USD 506.3 billion	3.7

Source: Hutton and Haller, 2004.

Finally, it is important to highlight that the estimates of health costs associated with inadequate service provision are limited to medical costs and losses in productivity, suggesting a potential underestimation of the total benefits (avoided costs) of improving WSS facilities. Moreover, the avoided health costs represent only one part of the total benefits of improving WSS facilities, omitting amenity, bathing, time saving, fisheries, tourism industries, and productivity gains in the agriculture sector, as well as the benefits from conserving natural resources.

Valuation of health impacts of air pollution

The epidemiological evidence related to air pollution is more uncertain than that for water pollution. PM appears to be the most health-damaging air pollutant, with well-recognised effects in terms of both morbidity and mortality. Not surprisingly, therefore, a number of valuation studies have focussed on WTP for reducing mortality risks associated with PM. The VSL estimates obtained from WTP range from approximately EUR 250,000 to EUR 1.5 million (Scapecchi, 2007 and Navrud, 2005)**.** However, there are significant differences, and these can be partly explained by differences in terms of incomes, the health-care system, culture, and experience with such surveys.

Navrud (2005) noted that one of the most controversial elements of estimating the health costs of inaction with respect to air and water pollution relates to the value placed on mortality. Table 11 reports estimates of the value of a statistical life (VSL) for six countries, based on the implementation of the same contingent valuation survey instrument related to air pollution, and showing values in the range of EUR 0.5–1.5 million. These values are also close to the interim central value of EUR 1.4 million that DG Environment of the European Commission used, which stemmed from an expert workshop organised by the European Commission DG Environment in 2000.[27] A review (Bellavance, Dionne and Lebeau, 2007)

26 Adjusted data provided directly by Hutton (personal communication, 16 March 2006).

27 However, Krupnick (2004, p. 32) noted that the European applications of the Krupnick *et al.* (2002) survey used the 5 in 1,000 risk change in 10 years (equivalent to a 5 in 10,000 annual risk change), but did not ask the 1 in 1000 WTP question first, as was done in the US and Canada. Based on the results in the two latter countries, he predicted that the implied VSLs for this smaller risk change would be 2-3 times larger than for the 5 in 1,000 risk change. The VLYs would be raised by a comparable amount. Krupnick also

of estimated VSL in different countries based upon wage risks studies[28] yields quite different (and generally higher) values.

Table 11. Value of a Statistical Life (VSL) Estimates, Using the Same Contingent Valuation Survey Instrument in Many Countries[1,2]

Country	Median WTP (2002 €)
USA	700,000
UK	772,000
Italy[3]	1,448,000
France	958,520
Brazil[4]	1,020,000 – 1,770,000

Source: Alberini *et al.* (2006) and Navrud (2005).

Notes: 1) Not adjusted for purchasing power parity (PPP)
2) Median values are reported here. The median value of the Weibull distribution is considered to be a more robust estimator. Mean WTP is 2-3 times higher, and should be used as upper end of the range estimate to show the uncertainty.
3) The relatively high Italian value may have been the result the Italian sample not being representative of the Italian population
4) The Brazilian study is based on a sample of middle and upper social class individual residents in Sao Paulo, roughly 69% of the total population (Ortiz *et al.*, 2004).

At the aggregate level, the health costs associated with air pollution can be considerable. AEA (2005) estimated that the 3.7 million life years are lost annually in those countries which now make up the EU-25, due to PM. This is equivalent to 348,000 estimated premature deaths. 21,000 deaths are also precipitated by O_3. The total health damages associated with prevailing EU legislation for O_3 and PM in 2000 for these same countries were estimated to be between EUR 276 and EUR 790 billion, with the mortality impacts from PM responsible for over two-thirds of these costs. In a more informal Swedish study, Huhtala and Samakovlis (2003) estimated that the total cost of NO_2 emissions may be as great as 0.7% of GDP. In a study of PM emissions in Singapore, Quah and Boon (2003) obtained an estimate of 4.3% of GDP. However, it must be emphasised that the estimated benefits are only relevant for marginal changes. As such, these figures are likely to be an over-estimate. Nonetheless, they do give an indication of the order of magnitude of health costs associated with air pollution.

Unfortunately, as in the case of water pollution, most valuation studies are based on the WTP for a given policy which results in an improvement in environmental conditions (and not the costs of inaction associated with the *status quo*). The fourth column in Table 12 provides estimates of the health benefits of policy interventions for a variety of air pollutants in different countries. The estimated health benefits are frequently considerable. Significantly, the costs are lower, and often significantly lower. Benefit-cost ratios in excess of five are obtained for diverse measures such as the use of low-sulphur fuels in Mexico, the CAFÉ programme in Europe, and NO_x control in Japan.

The estimates provided in the AEA (2005) study allow a comparison between the gross and net costs of inaction associated with not introducing policies which are more stringent than those presently included in the CAFE programme of the European Union. Under four different scenarios, one of which ("low") is less stringent than CAFE, it can be seen that gross costs of inaction (benefits of action) are rising, with a maximum of approximately $30 billion for the scenario in which all technologically feasible options are applied. However, the net costs of inaction (benefits of action) are negative for this scenario (Figure 9).

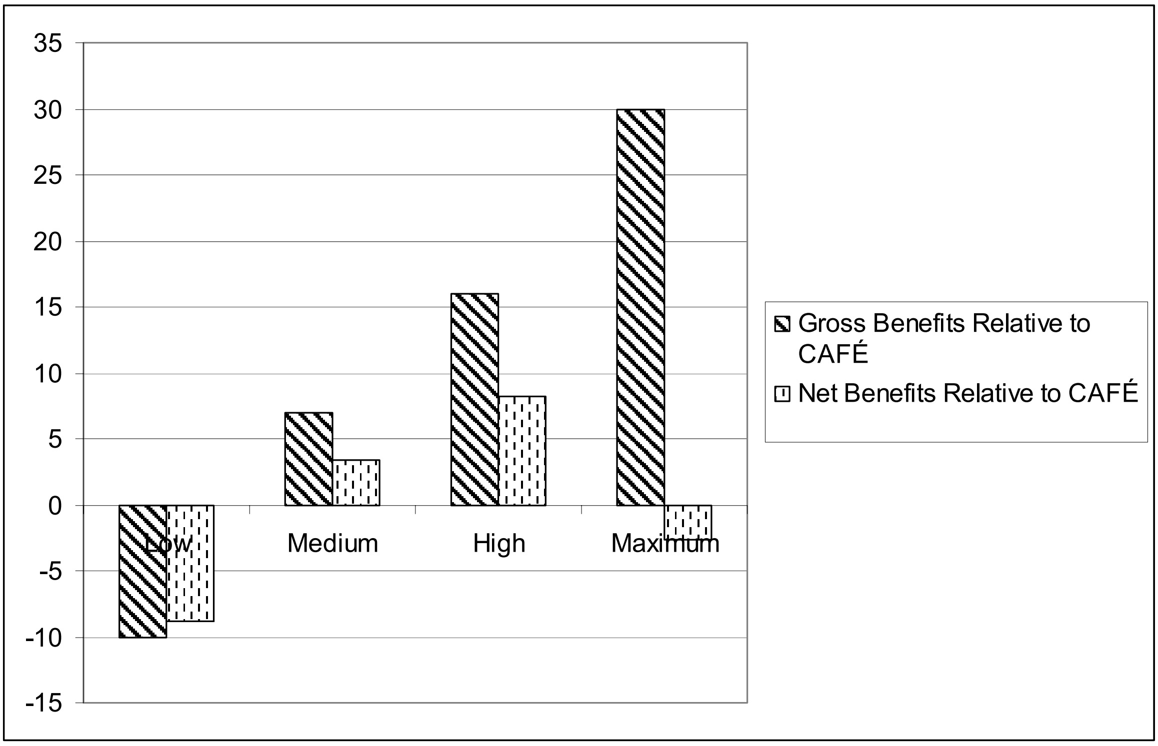
questioned the use of the median, even if it is a more "robust" statistic, arguing that the mean is the more appropriate measure to use in CBA, because it reflects the heterogeneity of values in the sample.

28 An approach which some OECD Member country governments advocate.

Table 12. Estimated Costs and Benefits of Policies Aiming at Improving Air Quality

Study	Country	Type of intervention	Health Benefits (Avoided Costs)	Implementation Costs	Net benefits	Benefit to cost ratio
AEA Technology (2005)	Europe	Current CAFE strategy	€42-135 billion	€7.1 billion	€34.9-127.9 billion	6-19
		Low reduction of air pollution	€37-120 billion	€5.9 billion	€31.1-114.1 billion	6-20
		Medium reduction of air pollution	€45-146 billion	€10.7 billion	€34.3-135.3 billion	4-13
		High reduction of air pollution	€49-160 billion	€14.9 billion	€34.1-145.1 billion	3-11
		Maximum technically feasible reduction of air pollution	€56-181 billion	€39.7 billion	€16.3-141.3 billion	1.4-4.5
Pandey and Nathwani (2003)	Canada	Reduce PM & O3 concentrations as follows: PM10 = 60 µg/m3; PM2.5 = 30 µg/m3; and, O3 = 65 ppb	CA$7,552 million	CA$2,491 million	CA$5,061 million	3
Blumberg (2004)	Mexico City and Mexico	Ultra-low sulphur fuels	Mexico City: US$2,456-US$4,874 million	Mexico City: US$120-US$250 million	Mexico City: US$2,206-US$4,624 million	10-19
			Mexico: US$9,665-US$12,083 million	Mexico: US$648-US$1,354 million	Mexico: US$8,311-US$10,709 million	7-9
USEPA (1999)	US	Reduce PM10, PM2.5, NOx, SO2, CO and VOC emissions	2000: US$71 billion	2000: US$19 billion	2000: US$52 billion	4
			2010: US$110 billion	2010: US$27 billion	2010: US$83 billion	4
Stevens *et al.* (2005)	Mexico	Reduce diesel-related PM emissions	Catalysed filters: US$0.8-2 million	Catalysed filters: US$0.2-0.4 million	Catalysed filters: US$0.4-1.7 million	2-5
			Active regeneration filters: US$0.8-2 million	Active regeneration filters: US$0.4-0.7 million	Active regeneration filters: US$0.1-1.4 million	1.1-3
			Oxidation catalysts: US$0.2-0.7 million	Oxidation catalysts: US$0.1 million	Oxidation catalysts: US$0.1-0.7 million	2-7
Voorhees *et al.* (2000)	Japan	NO_x control interventions begun in 1973	US$14,018 million	US$2,330 million	US$11,688 million	6
AEA Technology (2004)	UK	Air quality policies from 1990 to 2001	Road transport policies: £2,941-£18,370 million	Road transport policies: £2,000-£4,000 million	Road transport policies: £0.9-£14,370 million	1.5-5
			Electricity policies: £10,809-£50,609 million	Electricity policies: £2,000 million	Electricity policies: £8,809-£48,609 million	5-25
MFE (2004)	New Zealand	Introduce ambient and emission air quality standards	NZ$ 420 million (VSL only) NZ$ 9 million (cost of illness only)	NZ$ 111 million	NZ$ 318 million	3.87

Figure 9. Gross and Net Costs of Not Introducing Policies More Stringent Than CAFÉ (€ Billion)



Source: AEA Technology (2005).

Public policies should take all such costs into account, in order to maximise social welfare. However, it is important to distinguish between types of costs. The "costs" associated with pain and suffering are very different in nature from the "costs" of increased public health service expenditures. Depending on the institutional context, the actual "bearer" of these costs may also differ widely. In a study of respiratory problems from air pollution, Chestnut *et al.* (2005) distinguished between costs which are borne by the victim (except pain and suffering) and those borne by third parties (caregivers, taxpayers, *etc*). It is interesting to note how small the proportion of such financial and opportunity costs borne directly by the individual sufferer actually is (Table 13).

Table 13. Costs of Illness for Patient and Others (Acute Respiratory Problem from Air Pollution)

		$2002	%
3rd party costs	Total hospital charges	25,456	73.03%
	Post-hospitalization medical care	1222	3.51%
	Lost earnings for family/friends	427	1.22%
Individual costs	Out-of-pocket medical expenses	235	0.67%
	Out-of-pocket service expenses	238	0.68%
	Lost earnings for patient	4202	12.05%
	Lost household production	2669	7.66%
	Lost recreation value	409	1.17%

Source: Chestnut *et al.* (2005).

While this example (based on California data) provides an indication of the breakdown of the "costs of illness" by type of cost and bearer, institutional factors are also important. In this case, it is assumed that 75% of sick leave costs are borne by the patient, and that the patient only pays 3% of medical expenses.

These percentages will depend upon prevailing markets and policy factors. Such differences clearly affect the distribution of costs. Perhaps more significantly, the balance between costs of health services will vary widely across countries, with costs being borne by the patient (out-of-pocket or insurance premiums) or the taxpayer to very different degrees.

Perhaps most significantly, the health costs listed in Chestnut *et al.* (2005) do not include more "subjective" costs associated with pain and suffering. The relative importance of these costs for different environment-related health end points can be assessed based on two studies. In one case, Stieb *et al.* (2002) estimate the economic benefits of reducing acute cardio-respiratory morbidity associated with air pollution in Canada. Rabl (2004) provided recommended values of unit costs that should be used in France for different morbidity risks. These values are reported in Table 14. As can be seen, focussing on the costs of illness, without taking pain and suffering into account, can result in a gross underestimate, particularly for serious health impacts (*e.g.* cancer).

Table 14. % of Total Health Costs Related to Pain and Suffering

Health Endpoint	% attributable to pain and suffering
Stieb et al. (2002)	
Respiratory hospital admission	25.87%
Cardiac hospital admission	21.33%
Respiratory emergency department visit	46.73%
Cardiac emergency department visit	23.15%
Reduced activity day	47.92%
Asthma symptom day	57.14%
Acute respiratory symptom day	7.69%
Rabl (2004)	
Cancer, fatal	96.67%
Cancer, non-fatal	90.00%
Restricted activity day	37.69%
Workday lost	37.69%
Emergency room visit	32.27%
Asthma attacks, per case	14.09%
Asthma, per year	10.00%
Simple bronchitis	47.69%
Severe bronchitis	50.00%
Laryngitis or pharyngitis	51.67%
Sinusitis	33.33%

The discount rate is also critical to valuing the health impacts from local air pollution when the impacts are felt in the distant future (Hepburn, 2007). Long time horizons can be relevant for two reasons. First, some pollutants are persistent, remaining in the local environment for years, if not decades. Second, some of the health impacts of pollution occur from long-term exposure or are only experienced decades after exposure. As such, any assessment of the costs and benefits of air pollution policies necessitates trading off the costs of reducing pollution now -- against the benefits of better health (and more life years) decades into the future.

To provide a specific example, the health benefits of a policy to reduce fine particulates ($PM_{2.5}$) in England and Wales by 10 $\mu g/m^3$ from 2010 through to 2109 has been estimated using different discount rates: 6%

constant; 3.5% constant; and a declining discount rate starting at 3.5%.[29] As Table 15 illustrates, the discount rate adopted is enormously important to the final estimate of the health benefits (avoided health costs) of this policy. For example, moving from a 6% discount rate to the HM Treasury (2003) scheme would triple the relevant valuation of the benefits of the policy.

Table 15. Examples of the Impact of the Discount Rate on the Costs of Inaction with Respect to $PM_{2.5}$

Discounting scheme	1% $PM_{2.5}$ reduction
6% constant	GBP 31 billion
3.5% constant	GBP 82 billion
HM Treasury (2003)	GBP 93 billion

Source: Hepburn 2007.

Macroeconomic, labour productivity, and public finance implications of health impacts

Ill-health arising from environmental degradation can be a brake on the economy as a whole. In a macroeconomic panel study with data from 104 countries for 10 years, the effects of health on the economy were assessed econometrically. It was found that a one-year improvement in life expectancy contributes to a 4% increase in output (Bloom, Canning and Sevilla, 2001). In a panel of 50 countries over the period 1965-1990, Jamison *et al.* (2004) found that the adult survival rate accounted for approximately one-tenth of economic growth. The direction of causality in the health-growth relationship is, of course, important to assess. On the basis of American data, Brinkley (2001) found unambiguous evidence that poor health has a causal effect on wealth.

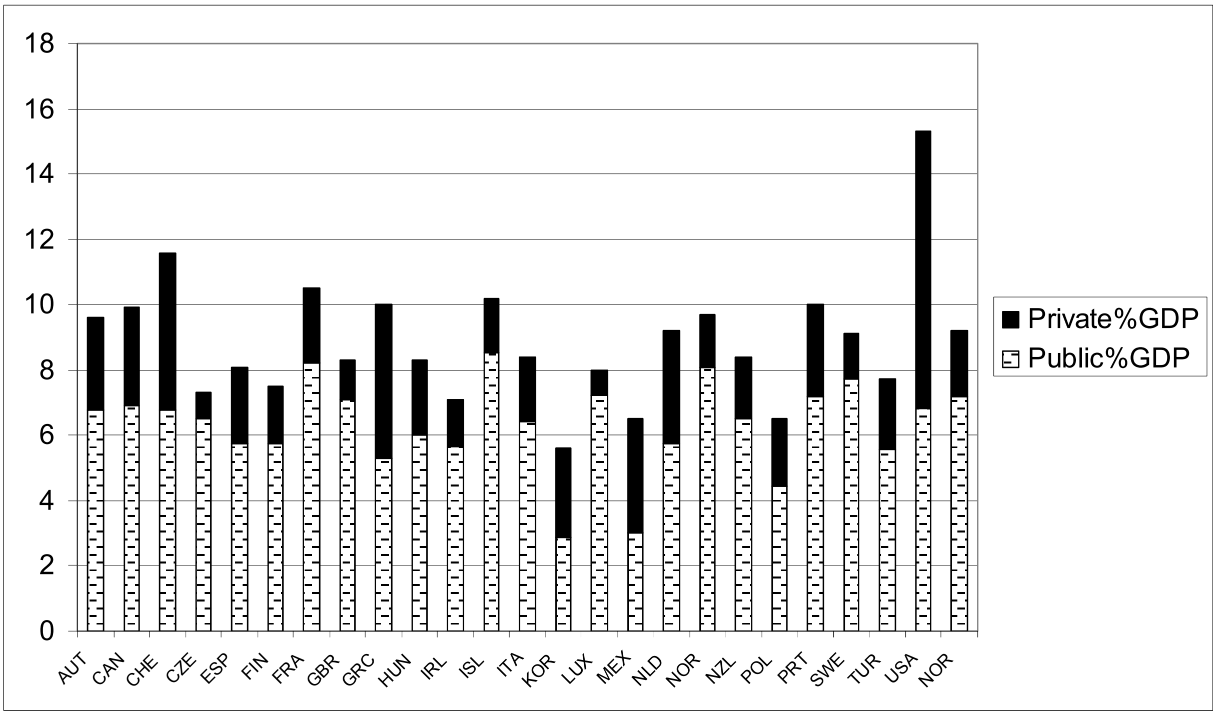
What drives these results? As noted above, the health impacts associated with pollution can have important implications for productivity. “Healthier workers are physically and mentally more energetic and robust. They are more productive and can earn higher wages. They are also less likely to be absent from work because of illness or illness in their family. Illness and disability reduced hourly wages substantially.” (Bloom, Canning and Sevilla, 2001).

There are also direct impacts on health service costs, and thus on public finances. Figure 10 presents data for 2004 on the percentage of GDP of total health service costs in OECD countries. In most countries, health expenditures represent between 6% and 10% of GDP, and these figures have been increasing. In the period 1997-2003, real health expenditures in the OECD grew by over 4% per annum (OECD, 2006b). However, Figure 10 also illustrates that there is variation in terms of who bears these costs. For instance, the percentage borne by the public sector (taxpayers) varies markedly. While the US has by far the highest proportion of health expenditures in GDP, Sweden, Iceland and Norway have the highest percentage of public health service expenditures in GDP.

In the UK, it has been estimated (UK Department of Health, 1999) that the total costs of respiratory diseases (GBP 566 million in 1996/97 prices) accounted for around 6% of National Health System hospital costs, and around 12% of the National Health System primary care expenditures. While environmental factors are clearly only one contributor to respiratory concerns, changes in pollution levels can have a significant impact on hospital admissions – a UK study found that a 1% reduction in the prevailing level of PM_{10} would result in a 0.14% reduction in respiratory hospital admissions (Maddison, 2004). A study by the Ontario Medical Association (2005) estimated that the healthcare costs associated with $PM_{2.5}$ and ozone in Ontario were $CDN 507 million *per annum*.

[29] The contribution of Brian Miller and Fintan Hurley (Institute of Occupational Medicine), Emma Powell (UK Defra), Heather Walton (UK Department of Health), and Paul Watkiss – in making available the data and research which underpins these results – is gratefully acknowledged. See also Chapter 4, this volume.

Figure 10. Public and Private Health Expenditures in the OECD (2004)



Source: OECD (2006b).

Health service costs are not the only impact which shows up directly in existing markets. For instance, the relationship between pollution, sick leave (or reduced activity days) and productivity has long been recognised. Samakovlis *et al.* (2004) estimated that an increase of 1 μg/m^3 in NO_2 emissions in Sweden resulted in a 3.2% increase in respiratory-related restricted activity days – approximately 685,637 additional restricted activity days. In a Norwegian study, Hansen and Selte (2000) found that the effect of reducing PM_{10} concentrations in Oslo from 24.5 μg/m^3 to 12.3 μg/m^3 would reduce the sick leave ratio by 7%. Earlier studies by Ostro (1983) and Hausman (1984) on the effect of TSP in the US found much greater impacts. (Table 16 summarises these results.) In the OMA (2005) study lost productivity costs are approximately $CDN 375 million *per annum.*[30]

Table 16. Effect of Pollution in Terms of Sick Leave and Restricted Activity Days

Study	Scenario	Effect
Hansen and Selte (2000)	Reduction in PM_{10} concentrations in Oslo from 24.5 μg/m^3 to 12.3 μg/m^3	Reduction in sick leave ratio by 7%
Samakovlis *et al.* (2004)	1 μg/m3 in NO_2 emissions	685,637 additional restricted activity days
Ostro (1983)	1 μg/m^3 increase in concentration of TSP in US	increase in probability of WLD in following two weeks of 0.13 percentage points
Hausman *et al.* (1984)	40% increase in TSP	10% increase in WLD

30 Total estimated costs (including pain and suffering and loss of life) were almost $CDN 8 billion *per annum*.

Non-health costs of air and water pollution

While this Chapter has focussed on the *health* impacts of air and water pollution, these are far from being the only costs associated with policy inaction in the area of air and water pollution. Many *non-health* costs are directly reflected in market prices. In a recent study of the Chesapeake Bay, Poor *et al.* (2007) found that a one mg/litre change (approximately 8%) in total suspended solids resulted in a fall in property prices of $1,086 (approximately 0.5%). For dissolved inorganic nitrogen, a one mg/litre change (300%) resulted in a $17,642 fall (approximately 9%) in property prices. A number of studies on the effects of water clarity on lakefront housing prices have been conducted in New England (Boyle *et al.* 1998, Michael *et al.* 1996 and Gibbs *et al.* 2002). Gibbs *et al.* (2002) found that a one-metre decrease in underwater visibility led to a decrease in property value of 6%. Similar results have been found for the effects of air pollution on residential property prices (*e.g.* Decker *et al.* 2005).

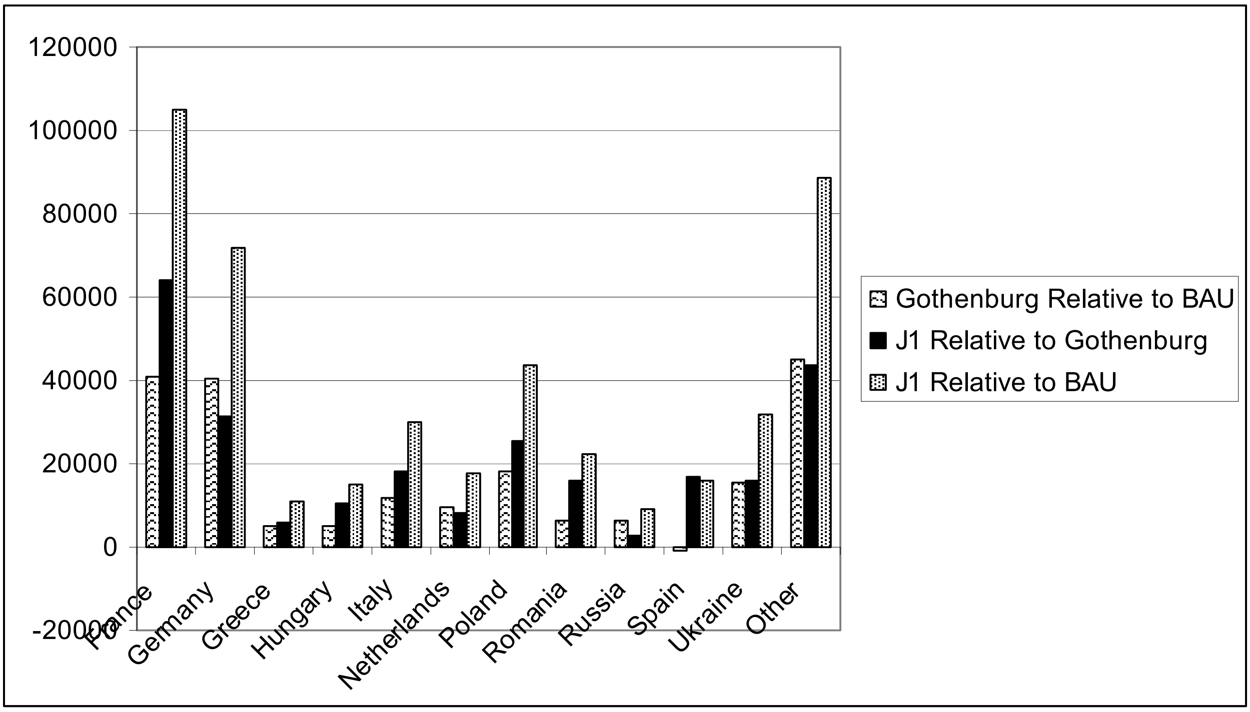
Other direct market effects include the effects on productivity in the agricultural, fisheries and forestry sectors. For instance, Shankar and Neeliah (2005) estimated that ground-level ozone concentrations in the UK were responsible for decreasing cereal yields by 1%, pointing out that the fall would be more significant if more tolerant seed varieties were not selected as an adaptive strategy. Kuik *et al.* (2000) estimated that a reduction in O_3 levels to natural background levels in the Netherlands would result in an economic surplus of EUR 310 million, with EUR 219 million accruing to consumers. Adams *et al.* (1986) reported that the gross benefits in terms of reductions of 10%, 25% and 50% in ambient O_3 concentrations in the US in 1980 would have yielded benefits of $0.7, $1.7 and $2.5 billion, respectively.

Holland *et al.* (2002) assessed the costs of inaction by comparing the effects of O_3 on agricultural yields in 2010 under three different policy scenarios: business-as-usual prior to the Gothenburg Protocol, full implementation of the Gothenburg Protocol, and a more stringent scenario (J1) which was considered in the negotiations of the Gothenburg Protocol. Assuming that "inaction" is the policy framework pre-Gothenburg, the gross costs of inaction relative to J1 are EUR 462 million per year. Taking Gothenburg as the counterfactual for "inaction", the gross costs of not introducing J1 were EUR 259 million per year. Figure 11 gives figures for selected countries.

Material damages can also be considerable. Lee *et al.* (1996) estimated the effects of 0_3 on surface coatings and rubber materials in the UK. They found that annual damages were in the region of GBP 170 to GBP 345 million per year. However, they recognise that there is considerable uncertainty about these figures. Olsthoorn *et al.* (1999) estimated that the materials damages in European cities associated with non-compliance of stationary sources with SO_2 air quality standards would be EUR 58 million per year. Interestingly, in their study of air pollution in Poland Dziegielewska and Mendelsohn (2005) found that the percentage of WTP for improved environmental quality (which is related to reduced materials damages) increased with the level of environmental quality.

AEA Technology (2005) has estimated the material damages associated with air pollution (mainly acidic deposition) in the EU-25 under current EU legislation. Natural stone and zinc-coated materials are the most affected. The estimated value of the damages was EUR 1.1 billion in 2000. Although very small, relative to the health costs obtained in the same study, these damages are far from negligible. Moreover, this excludes the costs of material damages to historic buildings and other sites of significant cultural heritage which are likely to be considerable, although exceedingly difficult to value in a meaningful manner (Navrud and Ready, 2002).

Figure 11. Costs of Reduced Agriculture Yields due to Ozone



Source: Holland *et al.* (2002).

Summary

The results of the studies cited in this Chapter give some indication of the costs of inaction with respect to air and water pollution. The costs arising through degraded ecosystems are important, affecting productivity in resource-based sectors, property markets and material damages. These have direct effect on the economy, even if the impacts are not always fully recognised. There are also other important costs, such as the impacts in terms of loss of biodiversity, some of which are important non-use values that are not reflected in market prices.

Health costs often represent 80% or more of total estimated social costs in valuation studies, particularly when important intangible costs such as those associated with "pain and suffering" are included. In addition, based on some assessments of the MDGs with respect to water supply and sanitation, the time and effort expended by households in non-OECD countries in order to gain access to unpolluted sources of drinking water are massive.

In addition to direct impacts on human welfare, these adverse health impacts have significant economic implications, including reduced labour productivity and adverse impacts on public finance. Overall, empirical evidence indicates that ill-health can be a significant drain on national economic performance. Therefore, to the extent that degraded environmental conditions contribute to ill-health, inaction with respect to air and water pollution is an issue of macroeconomic significance.

Most OECD countries have very well-developed policy frameworks to address concerns related to air and water pollution. "Inaction", in this sense, is a misleading term. However, the costs of not further improving environmental quality by setting more stringent policy objectives remain considerable. The results cited with respect to the CAFE programme and the Gothenburg Protocol are illustrative.

While this Chapter has focussed on the "costs of inaction" (*i.e.* the costs of not introducing more stringent policies than those which are in place), efficient policy interventions in the area of water and air pollution are dependent upon a careful balancing of both costs and benefits of "action". The costs of many policy interventions are also likely to rise steeply, as environmental objectives become more ambitious. Nonetheless, there are many areas in which the benefits far outweigh costs. In the case of non-OECD countries, improved WSH would appear to be one such area.

CHAPTER 3. COSTS OF INACTION WITH RESPECT TO CLIMATE CHANGE

Introduction

The costs of inaction related to emissions of greenhouse gases, and the associated problem of global warming, are potentially very large. Moreover, compared to most of the other environmental concerns addressed in this report, "inaction" with respect to climate change is relatively easy to conceptualise. Even though many countries have already introduced significant policy measures to curb GHG emissions, simulations indicate that the effects of policy measures introduced to this point will have only limited implications for GHG concentrations in the longer-term (OECD, 2008).

Using the JOBS/Linkage model a "baseline" scenario was run, as part of the process of developing the *OECD Environmental Outlook to 2030* (OECD, 2008). This scenario assumed that existing policies are maintained, and that no new policies are introduced. This is analogous to a "business-as-usual" scenario. On this basis, emissions of global CO_2 were projected to increase from 9.8 GtC in 2005 to 11.7 GtC in 2030. Figure 12 provides the resulting estimates, by gas and by sector.

Through their effects on GHG concentrations, these increased emissions will have implications for mean global temperatures. In the IPCC 4th Assessment (IPCC WG1, 2007), it was assumed that a doubling of carbon dioxide concentrations from pre-industrial levels (approximately 280 ppm) would lead to a temperature increase of somewhere between 2.0 ^{O} C and 4.5 ^{O} C, although there is some danger of much greater changes in global mean temperature.[31] Figure 13 provides the anticipated mean global temperature change, based on emissions (historical and projected) that emerged from the "baseline" scenario in OECD (2008), as well as the result obtained by assuming two related policy simulations (stabilisation at 450 ppm; and delayed introduction of a USD 25/ton carbon tax in 2020). In the former case, the associated tax would be about USD 100/ton carbon by 2040.

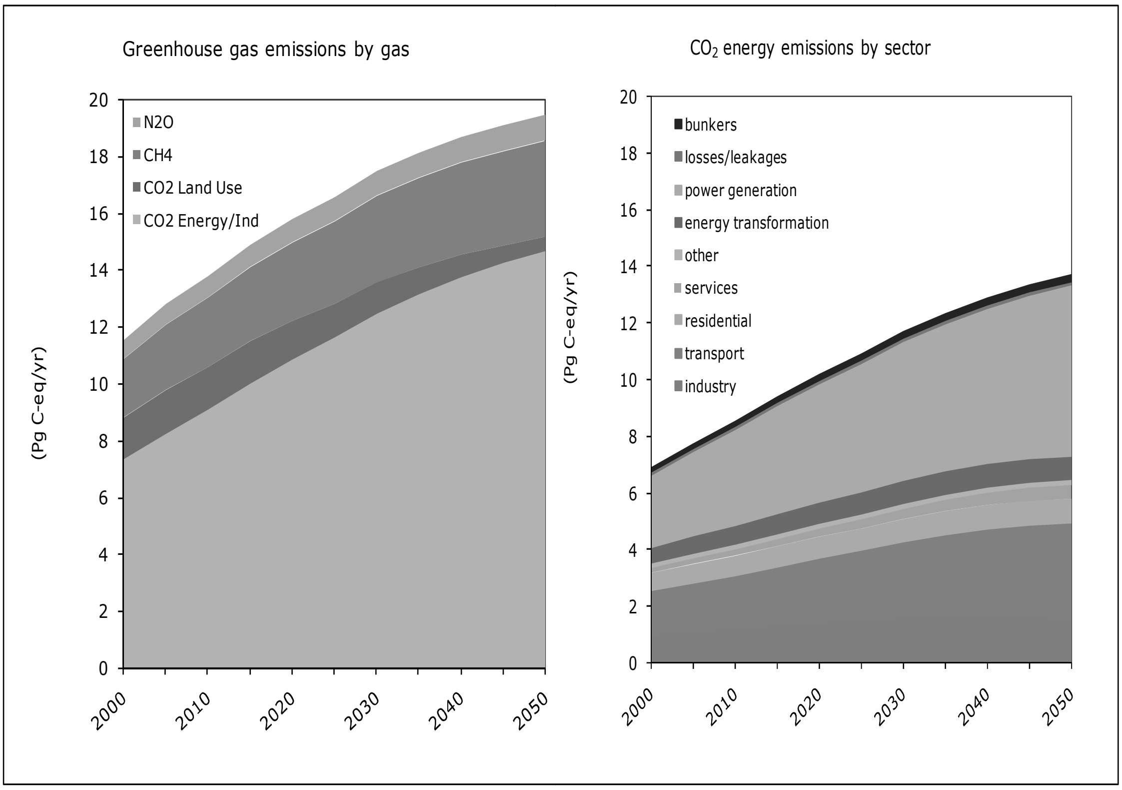
Extrapolating beyond 2050, these estimates are roughly in the middle of ranges of temperature change found in the peer-reviewed literature (IPCC WG1, 2007). For instance, the IPCC's "best-estimate" of the predicted 100-year global mean temperature increase ranges from 1.1 to 4.0 ^{o} C, with the full range of likely warming estimated to be between 1.1 and 6.4 ^{o} C.

The impacts arising from this increase in global mean temperature (and associated changes in precipitation levels) are likely to be significant. First, there will be significant market impacts on productive sectors, such as agriculture, forestry, and energy. There will also likely be a variety of market and non-market impacts on human health (*e.g.* diarrhoea, malaria, heat stress), as well as marine and terrestrial biodiversity. Extreme weather event activity, such as floods and hurricanes, is likely to increase. And finally, climate change might also lead to catastrophes, such as turning off thermohaline circulation in the North Atlantic, sudden and rapid release of methane emissions, or melting of the Antarctic or Greenland ice sheets.[32]

31 IPCC WG1 (2007).

32 IPCC WG2 (2007) gives an indication of the likelihood of major projected impacts.

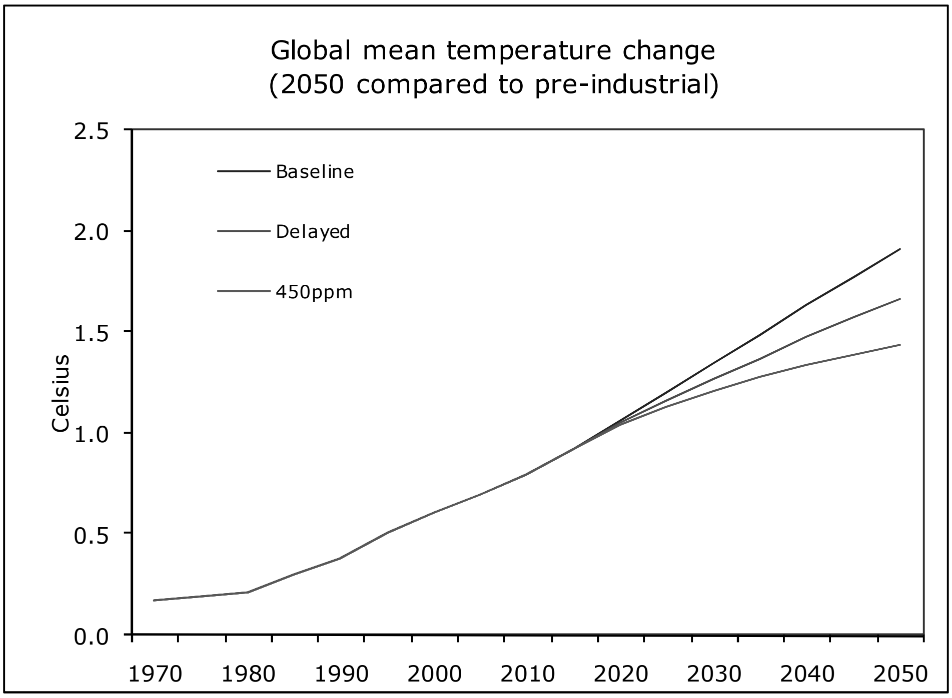
Figure 12. Estimated Emissions of Carbon Dioxide and Other GHGs[33]



Source: OECD (2008).

33 CO_2 emissions from energy (right panel) exclude CO_2 emissions from industry, which are shown in the left panel.

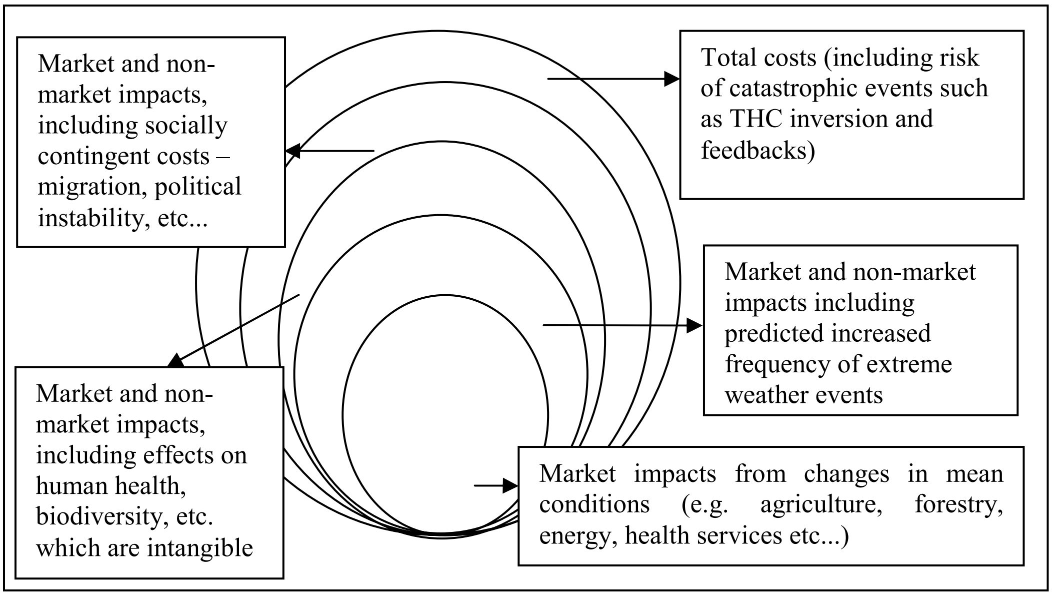
Figure 13. Global Mean Temperature Change, Relative to Pre-industrial Temperature



Source: OECD (2008).

The costs associated with such impacts are likely to be significant. These costs are represented in Figure 14. Most directly (and represented by the innermost circle of Figure 14), there will be significant market impacts. The outer circles add in other market and non-market costs associated with impacts for which there is increasing uncertainty. In general, the uncertainty associated with these different types of impacts increases, as one moves toward the outer circles.

Figure 14. Types of Costs Associated with Climate Change



Aggregate estimates of costs of inaction

There are two basic approaches to estimating the costs of inaction with respect to climate change:

- Estimates of the "social cost of carbon" (SCC). These are estimates of the damages associated with emitting an extra ton of carbon.

- Estimates of the "total economic damages". These are estimates of the damages associated with a given level of climate change relative to pre-industrial mean temperatures.

This Chapter draws on both approaches. It might be imagined that it is possible to derive estimates of total economic damages on the basis of the estimated social costs of carbon. However, estimates of the "social costs of carbon" are usually calculated on the basis of the estimated marginal damages from additional carbon dioxide emissions. With rising marginal damage curves, this will be an over-estimate. Second, the baseline used in the two estimates may not be the same. In many cases, the SCC is estimated with a view toward determining an optimal policy (*i.e.* where marginal benefits are equal to marginal abatement, mitigation, and adaptation costs). In the case of estimates of the total economic damages, on the other hand, the baseline often reflects "business-as-usual" conditions.

On the basis of a review of studies undertaken in the mid-late 1990s, Clarkson and Deyes (2002) concluded that the marginal social cost of carbon in 2000 is approximately GBP 70/tC ($100/tC). Reviewing this work, as well as some later studies, Pearce (2003) concluded that a figure of $4-$9 (GBP 3-GBP 6) would be more appropriate. Not all of this is difference can be explained by coverage of the studies. Part of the difference is attributable to the use of "equity weighting" in Clarkson and Deyes (2002). This issue is discussed in more detail later.

More recently, Tol (2005) reviewed 103 estimates of the SCC in the period 1991-2003 (Table 17). Including all estimates in an unweighted manner, the mean SCC was found to be US$ 93/tC.[34] Excluding all studies that were not peer-reviewed yielded a figure of $50/tC. The 5% and 95% confidence intervals give an indication of just how much uncertainty there is in these estimates. [35]

Table 17. Estimates of Marginal Cost of Carbon Dioxide Emissions ($/tC)

	Mean Estimate	5% CI	95% CI
Base	93	-10	350
Peer-Reviewed	50	-9	245
Source: Tol (2005).			

Formal estimates of the aggregate costs of inaction with respect to climate change are fewer in number, due to the significant modelling requirements associated with generating these estimates. Since the early 1990s, Nordhaus has produced a series of estimates based on the Dynamic Integrated Model of the Climate and Economy (DICE), the most recent of which are contained in Nordhaus (2007). His baseline scenario is one in which "no policies are taken to slow or reverse greenhouse warming". In other words, although individuals and firms may respond to climate change, governments do not introduce any new policies.

The discounted present value of damages for selected runs from the DICE model are provided in Table 18, with the baseline "no action" scenario generating estimated discounted aggregate environmental damages of US$ 22.65 trillion.[36] As a percentage of the discounted value of total future income, this is less than 1%. With a 50-year delay assumed in the implementation of "optimal" policies, the damages fall by approximately 20%, relative to the "no policy" scenario. Figures for the "optimal policy" (where estimated marginal control costs equal estimated marginal benefits), as well as a policy in which the temperature is constrained not to exceed 1900 temperatures by more than 2^{O} C, are also provided for comparison.

Table 18. Estimates of Aggregate Damages and SCC Under Different Policy Scenario Using DICE

	Aggregate (2005 US$ trillions)	SCC (2005 US$ per ton C)
No Policy[37]	22.65	30.7
50 Year Delay	18.68	30.5
Optimal Policy	17.19	29.8
Limit 2^{O} C	13.35	40.8

Source: Nordhaus (2007).

These figures are much lower than those reported in Stern (2007a), which were generated using the PAGE2002 Model. However, the metric employed by Stern ("per capita consumption equivalents")[38] is

34 Excluding estimates generated in order to replicate those derived from other studies.

35 There may be a bias in the peer-reviewed literature toward studies which have "incomplete" damage functions, since the treatment of market impacts is less controversial than the treatment of non-market and highly uncertain impacts.

36 In the model, the discount rate averages 4% over the course of the next century.

37 Actually a 250-year delay -- at which point the economy optimises emissions.

38 See Footnote 4 above.

different, so the results are not strictly comparable. Taking into account all potential impacts (market, non-market, extreme weather events, and catastrophic events), the discounted value of the costs of inaction with respect to climate change were estimated by Stern (2007a) to be 14.4% in the baseline "no policy" scenario. The social cost of carbon was $311 per ton C. A significant part of the differences between the Nordhaus (2007) and Stern (2007a) results can be explained by the discount rate used in the two studies.

Aggregate estimates of damages can also mask significant variation across countries. In his study using the FUND model, Tol (2002b) finds significant variation in the estimated economic cost of impacts across different regions. By 2200, Africa and Central and Eastern Europe are expected to bear damages equal to 8% of GDP, with Latin America and South and SE Asia experiencing damages of 5%. The Middle East and Centrally Planned Asian countries are actually estimated to benefit from climate change. However, there is broad agreement that the most significant impacts are likely to be felt in developing countries, because of their particular climatic conditions, the sectoral composition of their economies, and their more limited adaptive capacities.

Using temperature-damage relationships drawn from Tol (2002a and 2002b), a study by Kemfert and Schumacher (2005) produced higher figures for damage costs associated with a reference scenario in which no new climate policies are introduced. The total damage costs in 2100 represented 23% of global world output in 2100. The damages associated with "delayed action" are also assessed. In this latter case, no measures are undertaken until 2030, at which point measures are introduced to ensure that the increase in temperature is not greater than 2^{O} C. In this case, the damages in 2100 were equal to approximately 15% of world GDP.

Sectoral and Regional Estimates

There are a wide variety of potential damages arising out of climate change. This Section reviews estimates of these costs for four particular areas. Where possible, variation in the estimated impacts across world regions is also discussed.

Health Impacts

The health impacts of climate change (and increased climate variability) can be significant, including temperature-related illness and death (heat and cold), injuries and death from extreme weather events (such as floods and hurricanes), air pollution-related effects (such as respiratory problems), water and food-borne diseases (such as diarrhoea), vector and rodent-borne diseases (such as malaria and dengue fever), and food and water shortages, leading to malnutrition and dehydration.

The IPCC[39] summarised the main health impacts as follows:

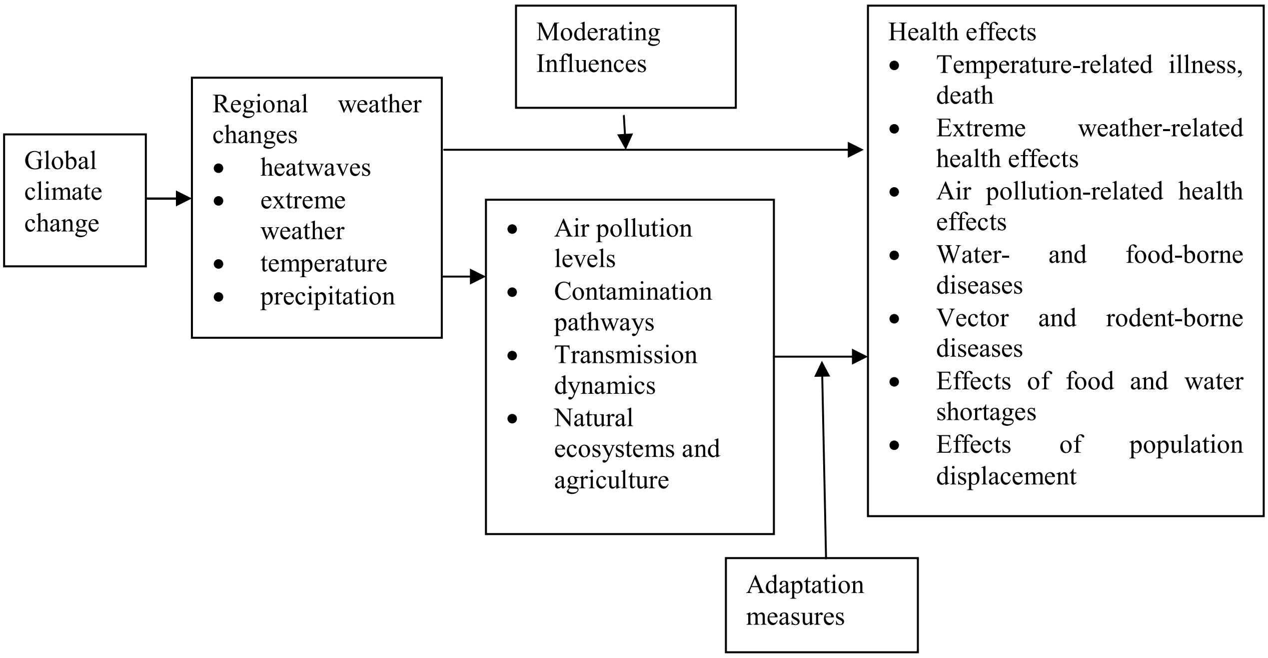
- Increases in malnutrition and consequent disorders, with implications for child growth and development;
- Increased deaths, disease and injury due to heat waves, floods, storms, fires and droughts;
- Increased burden of diarrhoeal disease;
- Increased frequency of cardio-respiratory diseases, due to higher concentrations of ground-level ozone related to climate change; and

[39] IPCC WG2 (2007) "Human Health" (Chapter 8).

- Altered spatial distribution of some infectious disease vectors.

While there may actually be some benefits associated with climate change in temperate countries -- due to reduced cold exposure -- these are far outweighed by the anticipated negative impacts. Figure 15 summarises the principal pathways.

Figure 15. Pathways from Climate Change to Health



Source: McMichael (1996).

There is already some evidence that climate change is affecting human health (WHO, 2003):

- 2.4% of world-wide diarrhoea was attributable to climate change in 2000;
- 6% of malaria in some middle-income countries in 2000 was attributed to climate change;
- human-induced climate change increased the probability of heat-waves, such as those in Europe in 2003, which caused an estimated 15,000 excess deaths.

While there is considerable uncertainty surrounding the health impacts being experienced at current levels of climate, McMichael *et al.* (2006) provide some rough estimates of the disease burden in terms of mortality and disability-adjusted life years in 2000 attributable to climate change (Table 19).

Table 19. Estimates of Burden of Disease in 2000 Attributable to Climate Change

	Mortality (000s)	DALYs (000s)
Malnutrition	77	2846
Diarrhoea	47	1459
Malaria	27	1018
Floods	2	193
CVD	12	NA
All	166	5517

Source: McMichael *et al.* (2004).

With climate change, this burden will rise, particularly in some regions. It has been estimated that the climate change-induced excess risk of different health outcomes will more than double by 2030 (Patz *et al*. 2005). For instance, the relative risk of diarrhoea in developing countries is predicted to increase from 1.01-1.02 in 2000, to 1.08-1.09 in 2030, resulting in an additional 47,000 annual deaths and 1.5 million DALYs. Ill-health from coastal floods is also predicted to rise markedly. Increases in the relative risk of malnutrition are also expected to be significant (McMichael, 2004).

Some risk factors (*e.g.* flooding) may increase almost by an order of magnitude. However, Campbell-Lendrum *et al.* (2003) show that some of the most significant impacts are associated with relatively small changes to risk factors, due to the relative importance of the baseline risk. Table 20 gives the 95% percentile estimates of the relative risk factors (by world region) for three specific problems in 2030, under three different temperature scenarios (McMichael, 2004).

Table 20. Percentage Increase in Health Risks in 2030 Due to Climate Change

Health impact	Diarrhoea			Death in coastal floods			Malaria		
Scenario	s550	s750	unmitig.	s550	s750	unmitig.	s550	s750	unmitig.
Africa – D	5	6	8	44	48	64	1	1	2
Africa – E	5	6	8	12	13	18	7	9	14
Americas – A	0	0	0	13	14	19	27	33	51
Americas – B	0	0	0	90	96	127	8	10	15
Americas – D	2	2	2	258	276	364	4	5	8
M. East/C. Asia - B	0	0	0	53	57	75	0	0	0
M. East/C. Asia - D	6	6	9	201	218	291	15	19	29
Europe – A	0	0	0	9	10	14	0	0	0
Europe – B	1	1	1	378	402	531	0	0	0
Europe – C	0	0	0	3	3	4	25	31	48
SE Asia – B	0	0	0	28	30	39	0	0	0
SE Asia – D	6	7	9	3	3	4	0	1	1
E. Asia/Oceania - A	0	0	0	3	3	4	25	30	48
E. Asia/Oceania - B	0	0	1	4	4	5	22	26	42

Note: The scenarios include: stabilization at 550 ppm, stabilization at 750 ppm, and unmitigated emissions.
Source: McMichael *et al.* (2004).

Sea Level Rise and Coastal Flooding

Sea level rise associated with global warming is attributable to thermal expansion of seawater, as well as the melting of glaciers, ice caps, and the Greenland and Antarctic ice sheets. The IPCC estimates of the relative estimated contribution of these factors in recent decades are summarised in Table 21.

Table 21. IPCC 4th Assessment Estimates of Contributions to SLR

Rate of SLR (mm per year)		
	1961-2003	1993-2003
Thermal Expansion	0.42+/-0.12	1.6+/-0.50
Glaciers and Ice Caps	0.50+/-0.18	0.77+/-0.22
Greenland ice sheet	0.05+/-0.12	0.21+/-0.07
Antarctic ice sheet	0.14+/-0.41	0.21+/-0.35
Sum	1.1+/-0.5	2.8+/-0.70
Observed Total	1.8+/-0.5	3.1+/-0.70

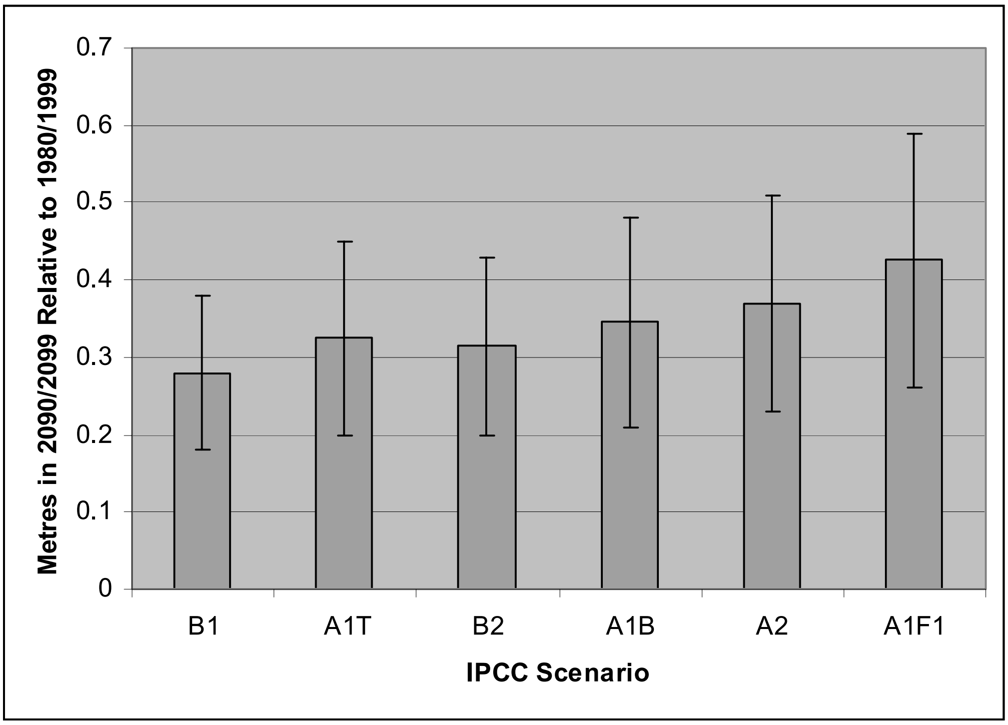
Source: IPCC WG1 (2007).

The IPCC also forecasts sea level rise in 2090-2099 to be as much as almost 60 centimetres above 1980-1999 levels -- if GHG emissions remain largely unconstrained. Even under the most stringent scenarios, the range of estimates is between 18 and 38 centimetres (Figure 16). However, the IPCC explicitly disregarded ice dynamics (*e.g.* changing migration of ice sheets), which may be particularly important in explaining the faster-than-expected rate of melting in Greenland.

Looking even further ahead, Nicholls *et al.* (2006) forecast potential sea-level rises which are much more dramatic. This is due in part to the slow thermal response of oceans, with on-going increases in thermal expansion for a considerable period even after stabilisation of concentrations. Combining impacts from thermal expansion, deglaciation of the Greenland and West Antarctic ice sheets, as well as melting of small glaciers, a sea-level rise of as much as nine metres is possible by 2500. However, the scientific uncertainty here is large, particularly with respect to the West Antarctic ice sheet.

Although the impacts of increases in sea levels are uncertain, they are likely to be significant. For instance, using IPCC 3rd Assessment scenarios, Nicholls and Lowe (2004) estimated that the loss of wetlands (relative to 1990) in 2140 will be as high as 20%-40% (when high climate sensitivity is assumed and impacts are unmitigated). Even with stringent mitigation efforts which stabilise concentrations at 550 ppm, losses under the “high climate sensitivity” assumption are between 10% and 30%. For the “medium climate sensitivity” assumption, the range is 2-12%. Nicholls and Lowe (2004) provided estimates of the number of people flooded in coastal surges under three different IPCC scenarios (unmitigated emissions, stabilisation at 550 ppm CO_2, and stabilisation at 750 ppm CO_2) through to the 2080s (Figure 17).

Figure 16. Projected Sea Level Rise Under Alternative IPCC Scenarios[40]

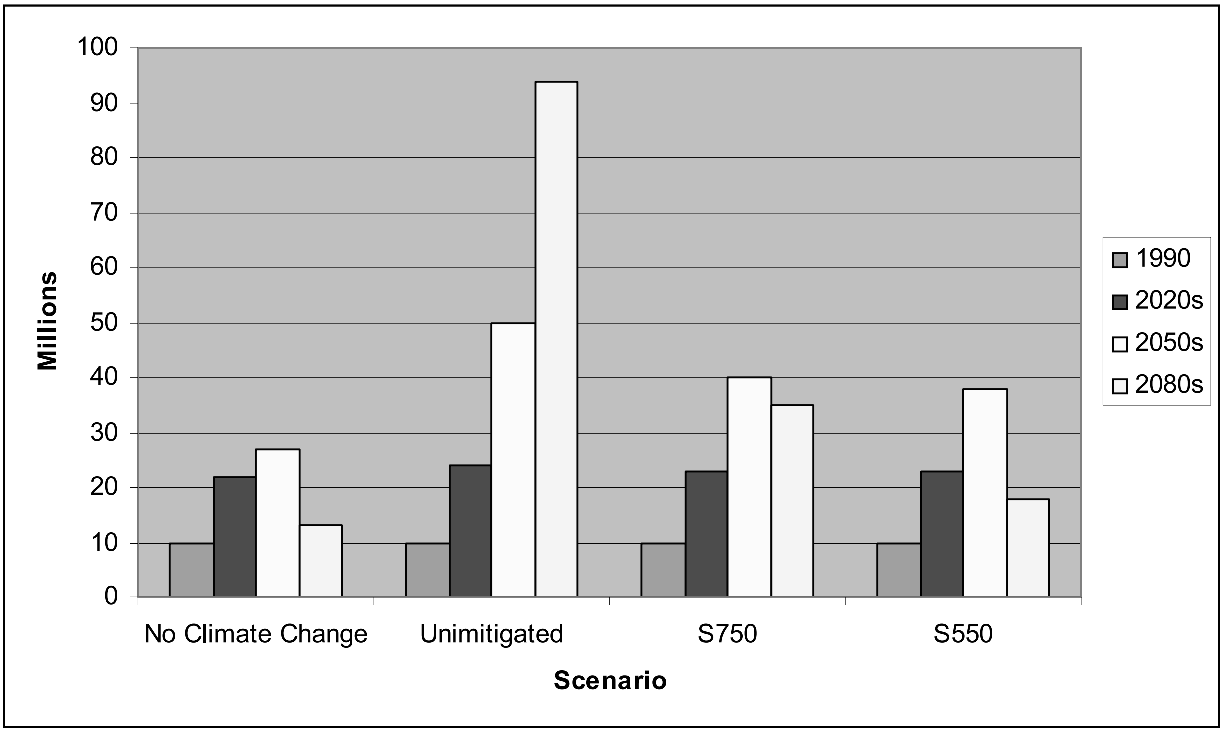


Source: IPCC WG1 (2007).

40 The IPCC scenarios can be summarised as:

- A1 – rapid and converging economic growth, inverse U population, and rapid technological change. Under A1F1 is technological change is fossil-fuel intensive, A1T involves the use of more non-fossil fuels, and A1B is balanced.
- B1 – is similar to A1, but with a structural shift in the economy toward services and an information-based economy.
- A2 – is autarkical, with little convergence in a fragmented global economy.
- B2 – involves considerable regional heterogeneity (growth and technology), with greater emphasis on environmental protection and social equity.

Figure 17. Number of People Affected by Coastal Surges in 2080s Under Different Policy Scenarios

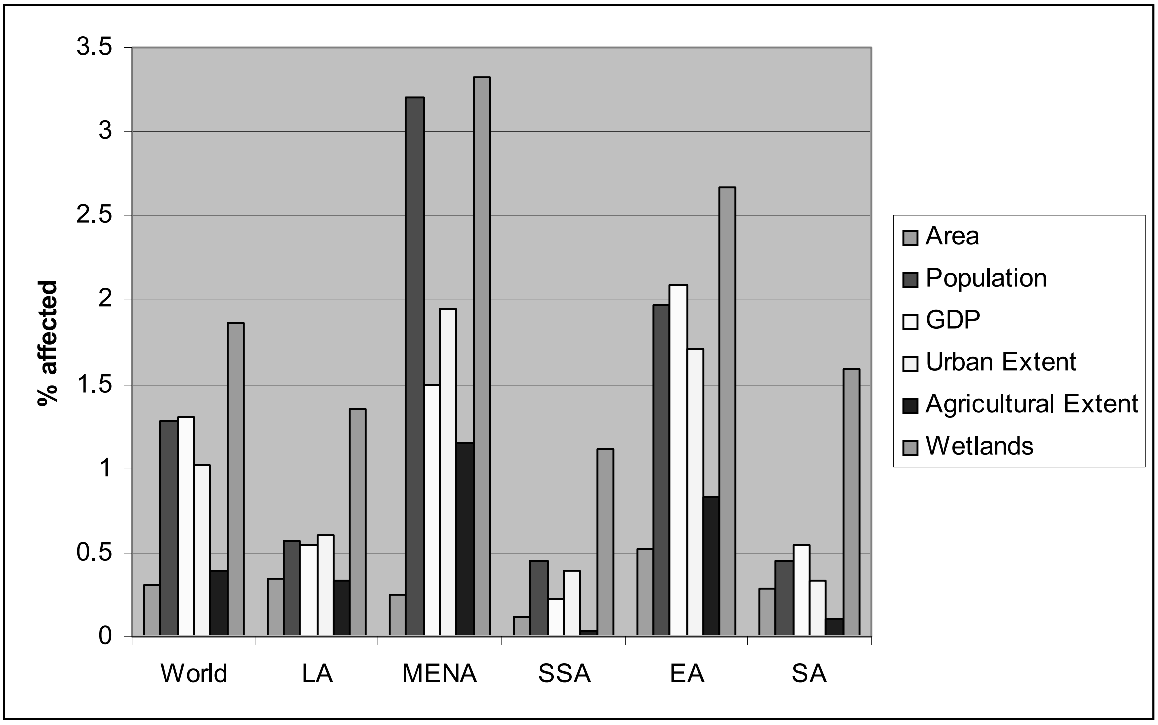


Source: Nicholls and Lowe (2004).

Dasgupta *et al.* (2007) present disaggregated impacts for 84 coastal developing countries, using GIS software and a variety of data sources. They found that, for a SLR of 1 metre, the most affected countries in terms of land area "affected" were the Bahamas (12%) and Vietnam (5%). For population, Vietnam (10%), Mauritania (9%), Egypt (6%), and Suriname (6%) were the most affected. In terms of GDP, the most "affected" countries were Vietnam (10%), Guyana (10%), French Guiana (8%), and Mauritania (8%). In terms of agricultural land, the worst "affected" countries were Egypt (13%), Vietnam (7%), and Suriname (6%).

Figure 18 presents the Dasgupta *et al.* (2007) results at the regional scale. The most affected region was Middle East/North Africa (MENA), with East Asia (EA) being the second worst-affected region. However, no adaptation is assumed in these models, and different capacities for adaptation may reverse these rankings.

Figure 18. Regional Impacts of 1 Metre SLR



Source: Dasgupta *et al.* (2007).

Agricultural Productivity and Food

Climate change can affect agricultural productivity in four main ways:

- *Changing temperature* – which can have a positive or negative effect on growing seasons at different latitudes;
- *Changing habitats* – which may increase or decrease the prevalence of pests and diseases to which crops are more or less vulnerable;
- *Changing precipitation patterns* – with increased risk of drought in some areas, and increased precipitation in others; and,
- *Changing carbon concentrations* – which can serve as a fertiliser for certain crops.

The net effect of these impacts will vary significantly by region and by crop. Generally, crop productivity will increase at higher latitudes (where higher temperatures lead to lengthened growing seasons, and sometimes greater precipitation) and decrease at lower latitudes (with increased heat and water stress). Estimated impacts on wheat are significant in North Africa, Central and West Asia. Maize yields are significantly affected in Western and Southern Africa, and in Latin America. In South and East Asia, rice yields are affected. For most crops and regions, carbon fertilisation accentuates the positive impacts and mitigates the negative ones. However, there is considerable uncertainty about the true impact of carbon fertilisation (Warren *et al.*, 2006a and 2006b).

Fischer *et al.* (2002) estimated that the impact of climate change on agricultural GDP in 2080 was between -1.5% and +2.6%, relative to the case where there is no climate change. However, there was considerable regional variation in these results. Comparing results using three different models, they found that, among developing countries, the number of countries which "lose" exceeded the number of countries that "gain"[41], and their decrease in cereal production was greater than gains elsewhere (Table 22).

Table 22. Developing Country Winners and Losers from Climate Change Impacts on Agriculture

	Number of Countries			Projected Population (billions)			Change in Cereal Production (million tons)		
	G	N	L	G	N	L	G	N	L
ECHAM4[42]	40	34	43	3.1	0.9	3.7	142	-2	-117
HADCM2[43]	52	27	38	3.2	1.2	3.3	207	3	-273
CGCM1[44]	25	26	66	1.1	1.1	5.5	39	3	-268

Source: Fischer *et al.* (2002).

Using the Hadley Centre's global climate model (HadCHM3), Parry *et al.* (2004) estimated the impacts of climate change on crop yield, production, and risk of hunger under the IPCC's main emission scenarios. Table 23 illustrates the change in crop yields, relative to the case in which there is no climate change -- for developed and developing countries. Developing countries were always worse off, relative to the baseline -- for example, the scenario with the highest CO_2 concentration showed a 7% decline for developing countries. For developed countries, yields actually increased under all scenarios, but the global effect was always negative, or (at best) neutral. Not only was there significant variation across countries; the implications for the risk of hunger also varied greatly, depending on assumptions made about the fertilising effects of increasing CO_2 concentrations.

Table 23. Estimates of Differences in Crop Yields due to Climate Change under Different Scenarios

Differences in average crop yield relative to baseline (in %) in the 2080s							
	A1F1	A2a	A2b	A2c	B1a	B2a	B2b
CO_2 (ppm)	810	709	709	709	527	561	561
World	-5	0	0	-1	-3	-1	-2
Developed	3	8	6	7	3	6	5
Developing	-7	-2	-2	-3	-4	-3	-5

Source: Parry *et al.* (2004).

41 Gainers (G) are countries with at least a 5% increase in cereal production, while losers (L) experience at least a 5% drop. Those in the range -5% to + 5% are neutral (N).

42 Max-Planck Institute for Meteorology and Deutsches Klimarechenzentrum.

43 Hadley Centre for Climate Prediction and Research.

44 Canadian Centre for Climate Modelling and Analysis.

Using the Basic Linked System world food trade model, Arnell *et al.* (2002) provided forecasts of the effects of climate change on cereal production for four different scenarios: no climate change; unmitigated (IS92a); stabilisation at 750 ppm by 2210; and stabilisation at 550 ppm by 2170. Assuming "no action" is taken with respect to emissions, positive changes in yields (due to warming, precipitation, and crop fertilisation) in mid and high latitudes were predicted to be more than compensated by reductions in the lower latitudes, particularly in Africa and the Indian sub-continent.

Changing crop yields (and demands) will affect market prices for agricultural output, as well as land prices. Mendelsohn and Williams (2004) incorporated these additional effects. Assuming "carbon fertilisation", they reported on the estimated impacts for a model with relatively low increases in CO_2 concentrations (CSIRO), and one with more significant increases (CGCM1). While global impacts in 2100 were positive in both cases, Asia and Africa suffered in both cases. Latin America also suffered under the CMCM1 scenario, but not under the CSIRO scenario (Table 24).

Table 24. Regional Climate Impacts in 2100 (Billion USD/year) – Cross-Sectional Estimates

	Estimates	
	CSIRO Climate Model	**CGCM1 Climate Model**
Latin America	2,3	-6,9
Africa	-2,2	-5,8
Asia	-14,5	-23,5
Oceania	2,7	1,2
N. America	11,9	8,0
W. Europe	9,6	9,0
USSR&EE	31,3	29,2
World	41,2	11,2

Source: Mendelsohn and Williams (2004).

Decreases in agricultural yields in developing countries are likely to have significant implications for risk of hunger. With temperature increases in excess of 3°C, and assuming no CO_2 fertilisation, there will be as many as 600 million additional people at risk of hunger in 2080, principally in Africa, West Asia, Latin America, and Central Asia (Warren *et al.*, 2006b). Even with CO_2 fertilisation, millions of additional people are likely to be at risk of hunger with temperature increases of 4°C.

Table 25 is drawn from the IPPC 4th Assessment.[45] Drawing upon two crop models (Fischer *et al.* 2005 and Parry *et al.* 2004), this Table compares the effects in terms of population at risk from hunger of different socio-economic scenarios and climate change ("with" and "without" carbon fertilisation), relative to the case where there is no climate change. Without carbon fertilisation, the increase in the number of people at risk could be more than 500 million people by 2080.

Arnell *et al.* (2002) also estimated that the likely increase in the population at risk of increased hunger in 2080 due to climate change was considerable. Relative to a "no climate change" baseline, in the "unmitigated" scenario, an additional 80 million people were affected. Even with stabilization at 550 ppm, an additional 40 million people would be at risk of hunger.

45 IPCC WG2 (2007), "Food, Fibre and Forest Products" (Chapter 5).

Table 25. Increase in People (Millions) at Risk of Hunger, Relative to Reference Case (No Climate Change)

	2050		2080	
	Fischer *et al* (2005)	Parry *et al.* (2004)	Fischer *et al* (2005)	Parry *et al.* (2004)
Climate Change with CO_2 fertilisation				
A1	11	2	28	28
A2	9	1	117	-27
B1	3	2	8	12
B2	-12	10	11	-12
Climate Change with no CO_2 fertilisation				
A1	NA	100	NA	262
A2	67	212	182	551
B1	NA	35	NA	35
B2	8	67	24	151

Source: IPCC WG2 (2007) "Food, Fibre and Forest Products" (Chapter 5).

Ecosystem Health and Biodiversity

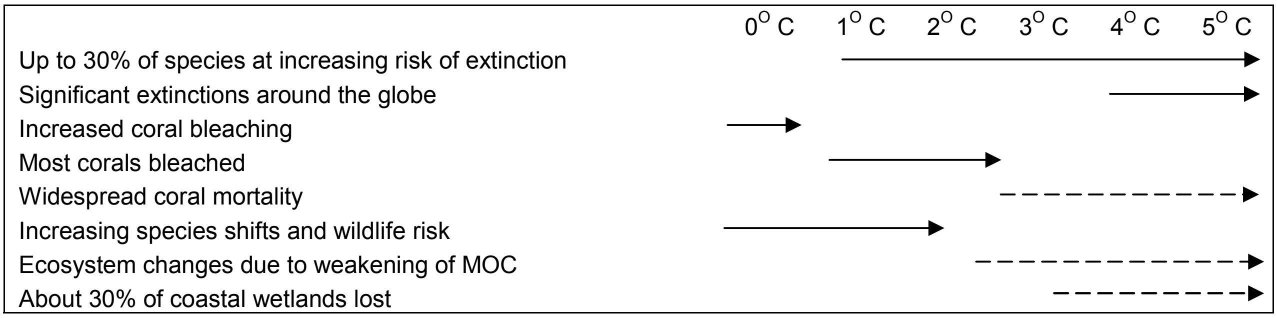
Ecosystem impacts are often not included in economic estimates of the costs of inaction of climate change. This is because there is considerable uncertainty about these impacts, based on difficulties associated with valuing these non-market costs. According to the IPCC,[46] both marine and terrestrial ecosystems are at risk from increased temperatures. Even with very low temperature increases (in the order of those already being experienced), there is evidence of coral bleaching and shifts in species habitat. In addition, there is high confidence that the extent and diversity of polar and tundra ecosystems is already in decline, and pests and disease have been spreading to higher latitudes and altitudes. Arnell *et al.* (2002) report the likely extent of "vegetation dieback" under the IPCC "unmitigated" scenario, (IS92a). In 2050, this comes to 1.5-2.7 million km^2, rising to 6.2-8.0 km^2 in 2080.

Figure 19 provides an overview of some of the areas for which there is reasonable confidence of likely impacts for different temperature increases. Globally, it is estimated that "net ecosystem productivity" would peak with a warming of 2^{O} C. Beyond that point, terrestrial vegetation is "likely to become a net source of carbon". It has been estimated that up to 43% of species in 25 biodiversity "hotspots" are at risk from warming in the region of 3-4^{O} C.[47]

[46] IPCC WG2 (2007), "Ecosystems, their Properties, Goods and Services" (Chapter 4).

[47] IPCC WG2 (2007), "Assessing Key Vulnerabilities and Risks from Climate Change" (Chapter 19).

Figure 19. Temperature Increases and Likely Impacts of Marine and Terrestrial Ecosystems



Note: Dotted lines indicate increasing impacts with temperature change.
Source: IPCC WG2 (2007) "Assessing Key Vulnerabilities and Risks from Climate Change" (Chapter 19).

With respect to marine ecosystems, there is some evidence that warming will result in reduced nutrient supplies for marine resources in the low latitudes, but increased productivity due to "light efficiency" in the high latitudes (Bopp *et al.*, 2001).

The damages to different ecosystems will largely depend on their capacity to adapt to changing climatic conditions, and on the rate at which the climate is changing. For instance, grasslands and deserts can adapt quickly, while forests will be slower to adapt (particularly at higher latitudes) -- not more rapidly than 0.05°C per decade (Arnell, 2006). Assuming a temperature increase of 2°C, Leemans and Eickhout (2004) estimated that more than 15% of the total area of ecosystems will be affected,[48] with 40% of this area being able to adapt. However, almost 20% of nature reserves will be affected, with less than 40% being able to adapt.

Warming is not the only climate change-related determinant of changes in ecosystems. Changing precipitation will also have important implications for ecosystem health and biodiversity, especially in Central Asia, Mediterranean, Africa, and Oceania (CBD, 2007). A small change in precipitation in desert ecosystems can also have devastating implications for local species.

Based on several previous studies which valued willingness-to-pay (WTP) for species, ecosystem and landscape preservation, Tol (2003) estimated the costs of ecosystem damages of a 1°C increase in temperature in different regions (Table 26). This amounts to 0.25-0.5% of world GDP. However, Tol acknowledged that these estimates are crude – due to uncertainties with respect to both impacts and valuation. For instance, Hitz and Smith (2004) pointed out that the evidence is not clear whether particular ecosystem impacts will be linear or exponential, with respect to increased warming. Similarly, WTP values are frequently transferred across regions, using methods which are (at best) approximate. Also, the study did not consider how damages change at higher levels of temperature increase.

[48] This is based on an indicator derived from the net change in extent of a particular ecosystem, expansion into other areas, and disappearance from existing areas.

Table 26. Estimate Costs of Ecosystem Damages

	Cost ($bn) of a 1^{O}C increase
OECD – America	-17.4
OECD – Europe	-14.7
OECD – Pacific	-11.5
CEE & FSU	-5.4
M. East	-0.3
L. America	-0.5
S&SE Asia	-0.1
CP Asia	-0.1
Africa	-0.1

Source: Tol (2002b).

Reasons for Variation

The previous Section has revealed the large variation in both aggregate and sectoral/regional estimates of the costs of inaction of climate change. Some of the reasons for this variation are explored below in more detail.

Incomplete Damage Functions

Differences in the estimates of the costs of inaction relate partly to the inclusion of different categories of cost. While all models include *market* impacts, many studies (*e.g.* Mendelsohn *et al.*, 2006) did not include *non-market* impacts, such as effects on biodiversity. Relatively few models include impacts associated with extreme weather events, with PAGE2002 (Alberth and Hope, 2006) being a notable exception. Low-probability catastrophic events are included in Nordhaus' (2007) DICE/RICE Model and Hope's (PAGE) Model. No models address the costs of social contingency effects (*e.g.* political instability), and it is difficult to foresee economic models doing so in a credible manner any time soon.

The effect of including a broader set of costs can be illustrated through results presented in Stern (2007a), using the PAGE2002 model (Table 27). With expanding coverage of cost categories (from just market impacts to the inclusion of the risk of climate-related catastrophes as well as non-market impacts), the estimated impacts increase from 2.1% to 10.9%. If climate-carbon cycle feedbacks are included, the central estimate rises to 14.4%.

Table 27. Estimates of Present Value of Environmental Damages

	% loss in terms of current consumption equivalents due to climate change[49]	5th percentile	95th percentile
Market Impacts	2.1	0.3	5.9
+ Risk of Catastrophe	5.0	0.6	12.3
+ Non-Market Impacts	10.9	2.2	27.4
+ Feedbacks	14.4	2.7	32.6

Source: Stern (2007a).

Even if GHG emission levels (and thus concentrations) could be forecasted with confidence, there is considerable uncertainty with respect to the effects of increasing GHG concentrations on different types of

[49] See footnote 39 above.

damages in different regions of the world. Even the shapes (let alone the positions and slopes) of the relevant damage functions are not known with certainty. In many cases, there is no simple (linear) relationship between concentrations, temperatures, and damages. In some cases, low levels of climate change may even provide *benefits* in terms of agricultural productivity for some crops in some regions. In other cases, negative and non-linear impacts may complement one another – thereby generating much greater impacts with even small increases in concentrations.

For instance, increasing sea surface temperatures may result in more-than-proportional increases in hurricane wind speeds; increased hurricane wind speeds may then result in more than proportional damages. On the basis of Australian insurance data, Hawker (2007) concluded that a 25% increase in peak wind gusts (from 40-50 knots to 50-60 knots) would result in a 650% increase in building damages. Similar (unstable) relationships are found for summer temperature-bushfire prevalence, and for precipitation levels-flood damages.

Valuing the costs of climate change-induced contributions to extreme weather events is particularly problematic. In addition to the inherent uncertainty associated with determining the extent to which the intensity of extreme weather events are affected by climate change, the macroeconomic impacts associated with a given event are difficult to assess. Using a model which allows for market rigidities in the adjustment to an extreme weather event "shock", Hallegatte *et al.* (2006) found that the overall impacts were much greater than if a smooth adjustment is assumed (as is the case in many models). Ultimately, with sufficient extreme weather event activity, an economy may find itself in "perpetual reconstruction", with the economic impacts being amplified. As Hallegatte (2006) pointed out, depending upon the timing: "the cost of two Katrinas would be much larger than twice the cost of one Katrina".

Given such uncertainty, it is not surprising to find that there is significant variation in the damage functions applied in different models. For instance, in the RICE/DICE (Nordhaus, 2007) and MERGE (Manne, 2005) Models, quadratic functions were assumed for market impacts. The PAGE2002 Model (Albreth and Hope, 2006) used linear or cubic functions. In the FUND Model (Tol 2002), a variety of functional forms were assumed for market damages, and in Mendelsohn's study (2006), hill-shaped functions were assumed for all market impacts -- an assumption that generally led to much lower estimates of the costs of inaction than the other authors generated.

In his runs of the PAGE2002 Model, Stern (2007a) produced damages based on temperature changes, relative to an increase of 2.5^{O}C, of the form:

$$Damages\alpha\left(\frac{T_R}{2.5}\right)^{\gamma}$$

In the baseline runs, the value of γ was assumed to have a mode of 1.3, a minimum of 1.0, and a maximum of 3.0. However, sensitivity tests run with the mode of γ ranging from 1.0 to 3.0 revealed considerable variation in the estimated costs of inaction. The effect of changing γ depends in part on the assumed marginal utility of income (μ), since this reflects the degree of risk aversion. Given the exponential shape of the damage function, higher values of γ yield much higher estimated costs (Table 28).

Table 28. Estimated Costs of Inaction, Assuming Different Damage Functions

	$\mu = 1$	$\mu = 2$
$\gamma = 1.0$	5.4	1.9
$\gamma = 1.5$	7.2	2.4
$\gamma = 2.0$	10.4	3.3
$\gamma = 2.5$	17.1	6.7
$\gamma = 3.0$	34.4	39.6

Source: Stern (2007b).

Ambrosi *et al.* (2003) investigated the effects of more complex "threshold damage functions", in which initial costs are limited, then rise very steeply at a certain point, before flattening out again. Relative to linear or quadratic damage functions, this resulted in relatively higher costs of inaction. Indeed, on the basis of their work, a case can be made for early abatement, even without accounting for catastrophic events.

There are many types of damages which cannot be adequately treated with a continuous and differentiable damage function -- including the costs of impacts which involve non-linear impacts (including threshold-based irreversibilities). Examples include:

- Collapse of the thermohaline current (THC) in the Atlantic Ocean;
- Release of methane emissions from thawing permafrost or warmer sea-beds;
- Switch of the El Nino/Southern Oscillation (ENSO) to a permanent state; and
- Deglaciation of the Greenland and Antarctic ice shelves.

In addition to the sheer magnitude of these impacts, accounting for such events in the estimation of the costs of inaction is complicated by the fact that they could occur suddenly and/or lead to irreversible changes. Moreover, the uncertainty associated with the timing and likelihood of these impacts is quite different from other types of uncertainty associated with climate change. While other types of impacts (*e.g.* effects on agricultural productivity, effects on human health) are not known with precision, probabilities can usually be attached to different possible outcomes for a given change in GMT. However, for events such as those listed above, the information is insufficient to posit a distribution of possible outcomes.

The degree of risk aversion in society is also important in valuing these impacts. There is good evidence to show that individuals perceive low-probability high-impact outcomes quite differently than they do other types of outcome. This suggests that estimating the degree of risk aversion is more complicated than simply carrying out certainty-equivalent assessments. Weitzman (2007) even suggested that these kinds of impacts should be treated separately in the estimation of the costs of inaction.[50]

Treatment of Adaptive Behaviour

An important issue in the determination of the costs of inaction relates to the treatment of adaptive behaviour. Pearce (2003) and Guo *et al.* (2006) argued that the treatment of adaptation goes a long way toward explaining observed differences in the estimated costs. Since initial estimates often assumed little or no adaptation, the effect of improved treatment of adaptation often had the effect of generating lower estimates of the costs of inaction.

[50] This kind of uncertainty is usually referred to as "fundamental", "hard" or "Knightian" uncertainty, after Knight (1921).

There are several different types of adaptation:

- *ecological* –the effect of changing climatic conditions on the location of ecosystems and species habitats;
- *physiological* – the effect of exposure to new diseases and pests on resistance (agricultural crops, human health); or
- *economic* –the effect of investments (such as dikes), output selection (such as crops) and input choice (such as fertilisers).

It is perhaps in the latter area that the most controversy has arisen. At the two extremes, one can distinguish between "pure myopic" behaviour (in which agents do not adjust at all in the face of a changing climate) and "perfect foresight" (in which agents anticipate all climate change, and adjust efficiently to it). In practice, actual behaviour will be between these two extremes, and there are a number of different factors which explain where on the spectrum particular agents are likely to be.

First, the rate of change in the impacts of climate change is important. Abrupt climate change is likely to result in less efficient adaptation (and thus, in higher costs) than more gradual climate change (Kuik *et al.*, 2005). This relates to the shape of the damage function. With exponentially increasing damages associated with a given change in temperature, adaptation is likely to be less efficient. This will be exacerbated if important thresholds are breached. The extreme case involves climate catastrophes (*e.g.* collapse of THC). Indeed, for Western Europe, THC collapse would imply adaptation first to warming, then to cooling.

Second, irrespective of the rate of change in climate impacts, there is likely to be variation in the cost of adaptation, depending upon the "fixity" and "longevity" of capital in different sectors (Nordhaus, 1999). With some infrastructure having a useful life of 100 years or more (bridges, tunnels, *etc*.), the loss of important "sunk capital", as a result of climate change, will be much greater than in situations where the capital turnover is more rapid.

Third, there are likely to be important information failures related to adaptation. For instance, it is difficult for agents to distinguish between temporary climate variability and permanent climate change (Callaway, 2004). On the one hand, if "variability" is mistaken for "change", agents may invest in costly adaptation that is inappropriate ("maladaptation"). On the other hand, if "change" is mistaken for "variability", agents may delay adaptation, incurring additional costs. The potential for error here is significant.

Fourth, even if change is accurately foreseen (and adaptation is efficiently undertaken), there may be important "learning costs" associated with such adaptation. For example, changing cropping patterns or the use of new seed varieties are likely to involve considerable trial and error. Adaptation to changing health threats is also likely to involve considerable "learning" on the part of public health services. Even the recognition and exploitation of changing tourism and recreation opportunities takes time.

And finally, there may be important market imperfections which constrain optimal adaptation. Migration is constrained (at least partly) by national borders, preventing optimal responses to changing employment conditions. Capital markets may be imperfect (and savings unavailable), constraining investment which would allow for the realisation of changing market opportunities. These imperfections and barriers reduce the potential scale of adaptation (Warren *et al.*, 2006a).

Although it cannot be assumed that agents are myopic, adaptation often takes place in the context of rapidly changing climatic conditions, with "lumpy" and "fixed" capital, and with imperfect information and imperfect markets. At least some of these conditions are of greater relevance to developing countries. Both Mendelsohn *et al*. (2006) and Warren *et al.* (2006a) argued that the poor are likely to be more affected than the rich. A number of factors are important in explaining this conclusion: more abrupt changes in poorer regions, fewer savings, underdeveloped markets, *etc*. In this context, policy initiatives to support adaptive action can be both efficient and equitable.

Estimates of the effects of adaptation on the estimated costs of inaction are not readily available, but some studies provide rough estimates:

- Fankhauser (2006) estimated that coastal adaptation (*i.e.* relocation and infrastructure) to sea-level rise could reduce the number of "vulnerable" people by 90%.
- Plambeck and Hope (1996) estimated that allowing for adaptation reduces the estimated marginal SCC by 50%.
- Nordhaus (1999) found that the costs of sea level rise are 50% higher if "myopic" foresight is assumed.
- Both Rosenzweig and Parry (1994) and Reilly *et al.* (1994) provided differentiated estimates for developed and developing countries. They found that the impacts on costs for the latter countries were much less significant than for the former ones.

Fankhauser (2006) summarised some of the studies which have been undertaken in the area of agriculture, differentiating by type of adaptation assumed (Table 29).

Table 29. Estimated Impact of Adaptation on Crop Yields

Study	Adaptation	Coverage	Impact Change
Easterling *et al.* (1993)	Planting date, tillage practices, crop choice, irrigation, drought-resistant crops	US Great Plains	29% - 60%
Rosenzweig and Parry (1994)	planting date, crop choice, irrigation	World	34% - 100%
Adams *et al.* (1993)	Planting date, tillage practices, crop choice, irrigation, drought-resistant crops	US	> 100%
Reilly *et al.* (1994)	Planting date, tillage practices, crop choice, irrigation, drought-resistant crops	World	39% - > 100%

Source: Fankhauser (2006).

Bosello *et al.* (2006) estimated the impacts of a rise in sea level of 25 cm, "with" and "without" adaptation. They also allowed for indirect (general equilibrium) impacts. They found that China and India will be the worst-affected, irrespective whether full or no adaptation is assumed. Japan is also very adversely affected, but there was a sharp difference in the latter case, depending upon whether there was adaptation.

There is significant potential for cost avoidance. In the presence of imperfect markets, imperfect information, and important distributional impacts, there is a clear case for public policy to support adaptation (not just to encourage mitigation). However, in order for such policies to reduce the costs of inaction, they must be targeted at particular agents, sectors, and regions where adaptation would not otherwise take place (or where it would be inefficient). It is clear from the list of types of adaptation considered in studies of the agricultural sector that some measures are likely to be undertaken autonomously by private agents (planting date, crop choice), even if this is not always done efficiently. Other adaptation initiatives will require some technical or support from public policy (*e.g.* investment in irrigation systems); while still others may require significant public sector involvement (*e.g.* new research on the development of drought-resistant crops).

Discount rates and intergenerational equity

Perhaps one of the most controversial issues in the assessment of the costs of inaction with respect to climate change is the choice of the discount rate. This is hardly surprising, given the very long- run nature of climate change impacts. The debate here has been given renewed prominence in the light of the Stern (2007a) report, in which a very low discount rate was applied.

According to the Ramsey Rule, the discount rate which should be used in the assessment of public programmes is a function of three factors: pure rate of time preference (ρ); the marginal utility of income (μ); and the assumed growth rate over the duration of project horizon (g). The *social* discount rate is then determined as follows:

$$SDR = \rho + \mu * g$$

Much of the discussion revolves around the choice of the private pure rate of time preference, which reflects "impatience". In most studies, a value of 1% to 3% is assumed. This is thought to reflect the "revealed behaviour" of agents in markets. Hepburn (2006) argued that, for policy concerns whose benefits and costs extend over centuries, there is no case for the use of a positive rate of pure time preference at all. Instead, he argued that a very low value on the pure rate of time preference (PRTP) should be applied, reflecting the "potential for extinction" (not "impatience"). In recent years, some analysts have also advocated for the use of *declining* discount rates – partly because of the uncertainty (or heterogeneity) of pure rates of time preference in the distant future (Weitzman, 2001).[51]

Differences in the choice of PRTP can have significant implications for the estimated costs of inaction. In his review of estimates of the marginal SCC, Tol (2005) provided estimates of the effects of differentiation in the PRTP. He finds that with a PRTP of 3%, the mean of the SCC is $16/tC (with a median of $7). With a PRTP of 1%, the SCC increases to $51/tC (with a median of $33). With a PRTP equal to 0%, the increase is to $ 261/tC (with a median of $39).

Using the PAGE2002 Model, Stern (2007a) favoured a value of 0.1, but also undertook sensitivity tests using higher rates. As can be seen in Table 30, the effect of changing the discount rate is significant, with estimated costs of inaction falling from 14.7% of per capita consumption equivalents when PRTP is equal to 0.1, to just 4.2% when a value of 1.5 is applied. Given other assumptions in the PAGE2002 Model, even the higher rate of PRTP only implies a discount rate of 2.8% – lower than the values used in most available studies.

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[51] This approach has so far only been adopted by a few OECD country governments in project evaluation.

Table 30. Effect of the Discount Rate on Estimated Costs of Inaction

PRTP	Discount Rate (%)	Discounted COI (per capita consumption equivalents)
0.1	1.30	14.70
0.5	1.80	10.60
1.0	2.30	6.70
1.5	2.80	4.20

Source: Stern (2007b).

Nordhaus (2006) compared “optimal” runs with the DICE Model, using the Stern assumptions, on the one hand, and his “best guess” assumptions, on the other. He started with the pure rate of time preference set at 3%, and then reduced it gradually to 1% over 300 years. In the latter case, the optimal carbon price in 2005 was $17.12/tC; in the former case, it was $159/tC.

Uncertainty with respect to the pure rate of time preference is, therefore, a significant source of variation in estimated damage costs. Unfortunately, uncertainty with respect to the choice of PRTP cannot be entirely resolved through more research. The difference between the two reflects a difference between the view that the choice of the pure rate of time preference is an empirical question, revealed in agents’ behaviour in the market (Nordhaus), and the view that the PRTP should be prescribed, and is therefore a political and ethical choice (Stern).[52]

Assumptions concerning μ (the marginal utility of income) also affect the social discount rate that is eventually applied. However, the role of this parameter in discounting is quite distinct from PRTP. Income or costs are discounted not specifically because of the position at which they occur in time, but because it is assumed that the world economy will grow through time. Indeed, with a decreasing marginal utility of income and negative growth, future income and costs would in fact be worth more (Azar and Sterner, 1996).

Using the FUND Model, Downing *et al.* (2005) reported that the value of the SCC increases three-fold, when μ is increased from 0 to 1. Evans (2005) reviewed revealed elasticities of the marginal utility of consumption in OECD countries, based on the structure of personal income tax rates. He found a mean value of approximately 1.4. This is within the range of 0.5-1.2 which Pearce (2003) argued was “reasonable”. Conversely, Dasgupta (2006) felt that this attaches too little importance to equity; he therefore proposed a value of between 2 and 4. However, such values are not consistent with observed distributional concerns in most countries.

The effect of the choice of μ on the social discount rate also has significant implications for the estimated costs of inaction. Table 31 gives Stern's (2007b) estimated damages under the “baseline” and “high” assumptions for μ (with climate-carbon cycle feedbacks) scenarios. An increase from 1 to 3 under the “baseline” scenario reduces damages by an order of magnitude. Under the “high” climate scenario, there is a U-shaped relationship -- with estimated damages increasing at higher values of μ. This latter finding can be explained by the dual role played by μ, both as a parameter of inequality aversion and risk aversion. Under the “high” climate scenario, the latter effect outweighs the former at high rates of μ.

52 Nordhaus (2006) pointed out that Stern’s choice of pure rate of time preference would require a marginal utility of income of 2.25 in order to reflect observed macroeconomic trends (savings, investment). With Stern’s assumptions for growth, this would increase the discount rate by a more than a factor of 2.

Table 31. Effect of Elasticity of Marginal Utility of Income of Costs of Inaction

Elasticity of marginal utility	% loss in per capita consumption equivalents due to climate change	
	Baseline	Feedbacks
1.0	11.1	14.7
1.5	6.5	10.2
2.0	3.6	7.4
2.5	2.1	8.1
3.0	1.3	13.2

Source: Stern (2007b).

In estimating the costs of inaction, it is the combined effect of the choice of parameters for the pure rate of time preference and for the marginal utility of income which will be most important, since together they will give the social discount rate for a given level of assumed growth. Because there is considerable uncertainty about both of these parameters, there is considerable uncertainty about the appropriate social discount rate to apply overall. Weitzman (2007) showed that the very uncertainty concerning the appropriate discount rate implies that it should be declining through time, converging toward lower values.

The "declining discount rate" approach was the basis of the framework proposed by the UK Treasury in their *Green Book* (2003). Based on sensitivity tests using the FUND model, Downing *et al.* (2005) reported that using such a framework produces estimated social costs of carbon which are not dissimilar to those which would be obtained using a PRTP of 1% (Table 32).

Table 32. The Effect of the Pure Rate of Time Preference on the Estimated SCC

PRTP	Best Guess	Average
Green Book (1.5% and declining)	£19	£24
PRTP = 0%	£56	£171
PRTP = 1%	£11	£43
PRTP = 3%	£-2	£-1

Source: Downing ***et al.*** (2005).

Equity and distributional issues

A further source of debate in estimating the costs of inaction relates to the treatment of distributional equity. As noted earlier, there is significant variation in the regional impacts of climate change. For most models, some regions even benefit – at least in the short-term. However, other regions suffer, even with very mild warming. Ultimately, all regions suffer significant net damages in most models, although there is significant variation in the specific burdens that are imposed.

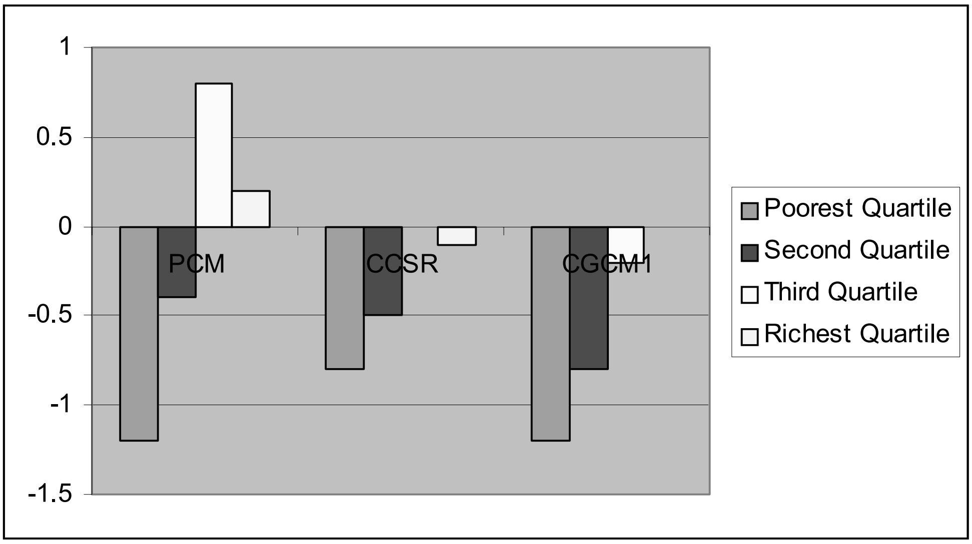
There is little question that the most significant impacts are likely to be felt in developing countries, because of their particular climatic conditions, the sectoral composition of their economies, and more limited adaptive capacities. The earlier review of impacts with respect to health, sea level rise, agricultural yields, and ecosystem damages supports this conclusion. At a more disaggregated level, Downing *et al.* (2005) identified three types of countries in particular which are most vulnerable:

- *Coastal deltas,* where dense populations are subject to increased coastal erosion, recurrent storm surges and cyclonic risk, with Bangladesh being the archetypical example;

- *Semi-arid regions,* where increased water stress will put further strain on marginal agricultural/pastoral systems, with the countries of the Sahel being archetypical examples; and
- *Small-island states*, where sea level rise and cyclonic risk threaten populations, perhaps inundating whole islands, such as in the South Pacific.

In their analysis of the distributional impacts of market damages from climate change, Mendelsohn *et al.* (2006) found that there is a distinct tendency for the poorest quartile (in terms of country populations) to bear the greatest burden, irrespective of the climate model selected (Parallel Climate Model, Center for Climate Research Studies, and Canadian General Circulation Model) (Figure 20). There is, however, some ambiguity for the third and richest quartiles, with two of the models (PCM and CCSR) showing *progressive* impacts. Sensitivity tests in the same study reveal that the distribution of damages is primarily a function of climatic conditions (and thus damages) -- not differences in economic structure.

Figure 20. Market Impacts as % of GDP in 2100



Source: Mendelsohn *et al.* (2006).

Recognising that the value of given absolute income losses (and income gains) is greater for those living at the edge of subsistence than for those with high incomes, some studies weight the costs (and benefits) of impacts accordingly.[53] Assuming that average world income is $\overline{Y}$, and Y_i is income in country i, the value of ε will determine the extent to which income will be weighted such as to attach greater importance to damages borne by the poor:[54]

$$D_{world} = \sum_{i}^{n} D_i * \left[\frac{\overline{Y}}{Y_i} \right]^{\varepsilon}$$

Under standard assumptions, weighting utility across different income classes, within a given generation, can have significant implications for the estimated costs of inaction. Stern (2007b) informally estimated

53 The use of weighting in project evaluation is so far only used by a few OECD country governments.

54 There is, not surprisingly, a close relationship between the marginal utility of income and the application of equity weights (Pearce, 2003).

that the effect of the application of equity weights would increase his estimate of the costs of inaction from 14.4% of per capita consumption equivalents – to approximately 20%. Pittini and Rahman (2004) and Pearce (2003) argued that equity weighting can *double* the estimated costs of inaction, relative to unweighted values. Table 33 provides estimates of the effects of weighting a given estimate of global damages ($322 billion), using different values of ε.

Table 33. An Example of the Effects of Equity Weighting on the Costs of Inaction

	PCM
Unweighted	$322 billion
ε = 0.5	$307 billion
ε = 0.8	$343 billion
ε = 1.0	$390 billion
ε = 1.5	$600 billion

Source: Pearce (2003).

Based on an extensive review of 103 studies, Tol (2005) gave an indication of the probability distribution of the marginal SCC ($US/tC), "with" and "without" equity weights. While the mean was similar, the median was markedly higher when equity weights were applied (Table 34).

Table 34. SCC "With" and "Without" Equity Weighting

	Mean	Median	5%	95%
No equity weight	90	10	-8	300
Equity weight	101	54	-20	395

Source: Tol (2005).

Summary

The potential costs of inaction with respect to climate change are considerable. Some of these costs are already being felt in both OECD and non-OECD economies. If emissions remain unmitigated, there is little question that the magnitude of the costs of climate change will prove to be very significant, disrupting economies in a manner unlike other types of environmental impact. Indeed, even if emissions are mitigated significantly in the very near future, the stock nature of GHGs is such that significant costs will be incurred due to past inaction.

Due to the potential magnitude of these impacts, the tools used for assessing the costs of policy inaction (and benefits of policy action) should be reviewed very carefully. Standard methods used in cost-benefit analysis assume that the basic structure of the economy remains unchanged over the period of the analysis. This allows the benefit-cost comparison to be made in terms of (small) marginal increments. For many types of environmental policies and impacts, this is appropriate – because it allows the analyst to circumscribe the impacts which need to be considered and the benefits which need to be valued, since the broader impacts on the economy will be trivial.

However, "climate change policy" is likely to have *non-marginal* impacts. Policy decisions concerning the climate have the potential to shift the entire trajectory of economies, with the macroeconomic context turning out to be very different under different scenarios. At the technical level, this means that some of the assumptions often used in valuing the costs of inaction are likely to be inappropriate. For example, Weitzman (2007) and Hepburn (2006) have pointed out that, if the discount rate that is used has important implications for the trajectory of the economy,[55] it will be inappropriate to compare the costs of different scenarios with different discount rates along a given trajectory of the economy.

Clearly, this is most relevant for impacts such as climate-related disasters. However, the point is more general. In one of the few studies to look at the effects of climate change on important macroeconomic fundamentals, Fankhauser and Tol (2005) undertook simulations which took into account the prospect of future damages on capital accumulation and savings rates. They found that, under plausible assumptions for different parameter values, these "indirect" costs can exceed the "direct" costs of climate change – with the difference becoming greater over time. Hallegatte (2006) pointed out that, in the face of rigidities in capital and labour markets, the costs will be greater still.

Perhaps more fundamentally, there is considerable non-probabilistic uncertainty about all of the cost estimates discussed in this Chapter – with some of the potential impacts likely to result in very significant economic consequences. Dietz (2006) argued that, in such contexts, standard cost-benefit analysis may not be appropriate, and that it may be better to approach the issue in terms of "safe minimum standards". Implicitly, this is consistent with the UNFCCC's Article 2, which calls for stabilisation of GHG concentrations in the atmosphere at a level that would prevent "dangerous anthropogenic interference". However, alternatives to the use of CBA through an integrated assessment model have their own weaknesses as well.

Perhaps the most important conclusion here is, therefore, that assessments need to be undertaken (and results presented) in a manner which takes due account of the uncertainties involved. At the simplest level, this includes the presentation of results with a range of assumed parameter values (sensitivity analysis). This approach may illustrate differences in the estimated costs of inaction which vary by an order of magnitude. It is this variation itself (and not just the central estimates) which should inform decision-making.

55 Formally, the discount rate is endogenous.

CHAPTER 4. COSTS OF INACTION WITH RESPECT TO ENVIRONMENT-RELATED INDUSTRIAL ACCIDENTS AND NATURAL DISASTERS

Introduction

This Chapter reviews the costs of inaction with respect to "environment-related" industrial accidents and natural disasters, which are (at least partly) induced by human activities. While this potentially covers a broad range of types of impacts, including floods, hurricanes, oil spills, nuclear accidents, and contaminated land, the focus here is only on "industrial" accidents, such as oil spills and contaminated land, as well as natural disasters such as hurricanes, floods and extreme weather events. Before proceeding to a discussion of the *specific* cases, however, this Introduction provides an overview of some of the general issues involved in assessing the costs of inaction with respect to environment-related industrial accidents and natural disasters.

In broad terms, such incidents differ according to:

- the relative importance of anthropogenic contributions to the probability of the industrial accident or natural disaster;
- the degree of irreversibility of the damages arising from the industrial accident or natural disaster; and
- the extent of unpredictability or uncertainty related to their frequency, timing and severity.

Due to the irreversible nature of many of these impacts, remediation (no matter how thorough) can only "recover" part of the benefits which previously existed. Once damages have arisen, it may therefore be difficult to justify the considerable *ex post* costs needed to remediate the problem. In effect, the damage costs are "sunk". Another important aspect of environment-related industrial accidents and natural disasters is that in most cases it will not be possible to reduce the probability of their occurrence to zero, with costs rising sharply as the probability diminishes.

Conceptually, the risk posed by a given event (*i.e.* the potential damage inflicted) can be expressed as a function of: hazard rate (a probability distribution characterized by the frequency, intensity, and location of an event); and vulnerability (the capacity of the community to withstand such hazards, resulting from physical, social, economic, and environmental factors – see for example, Nordhaus, 2006). More specifically:

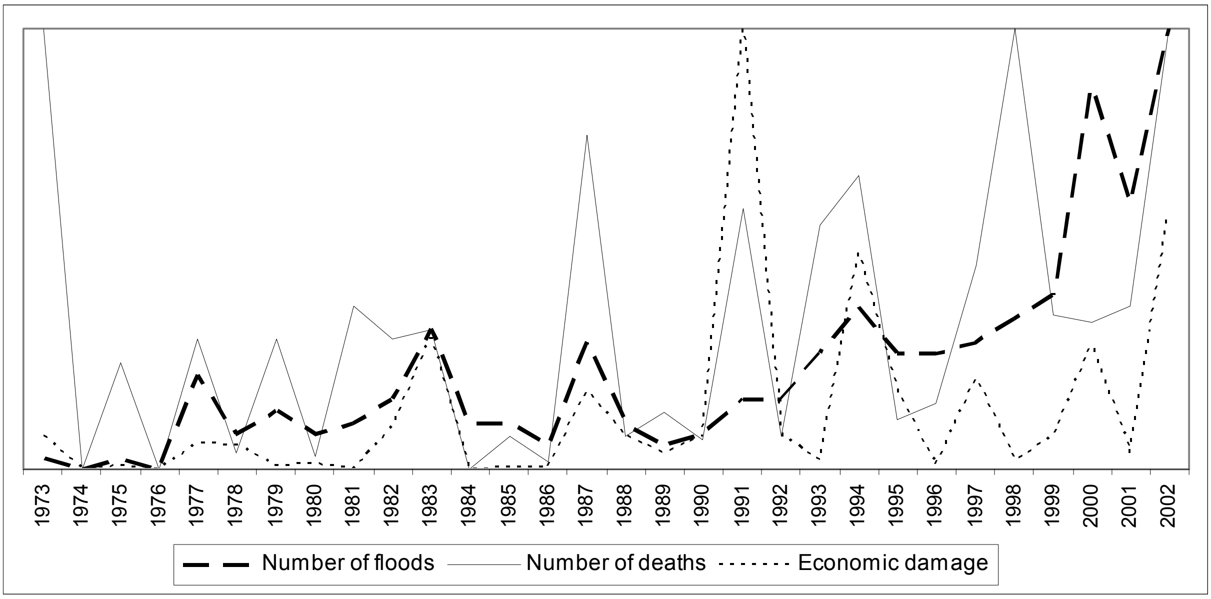
$$\text{Risk} = f\,(\text{Hazard rate} \times \text{Vulnerability})$$

Damages therefore arise from the interaction between natural conditions and vulnerability. This simplified model suggests that there are two basic avenues through which damage can be minimized by policy interventions. The first case targets the "hazard rate". In the case of environment-related industrial "accidents" (such as oil spills), the hazard rate is partly a function of preventive behaviour on the part of those agents who are potentially responsible for any adverse impacts which might arise. More indirectly, for environment-related natural disasters, the hazard rate is linked to anthropogenic factors, such as local conditions (availability of adequate flood plains) and/or global factors (such as climate change impacts or the occurrence of tropical storms and extreme temperature spells). This implies that, even if it is not possible to entirely prevent a specific event, preventive measures can reduce the probability of it occurring. If the likelihood of an event occurring is at least partly related to anthropogenic factors (*e.g.* CO_2 emissions, safety measures in the maritime fleet), in the longer term, the frequency and severity of future events can be affected by reducing the hazard rate.

The second set of strategies for risk management targets "vulnerability". The vulnerability of a community is closely related to factors such as economic geography (location of settlements and economic activity in relation to the affected areas), as well as the overall volume of economic activity (population size, value of productive assets, capital intensity of output). Disaster mitigation strategies should thus aim at: reducing the impact of an event should it occur (*e.g.* construction of water levees and dikes); and increasing community preparedness (*e.g.* infrastructure for information dissemination, traditional emergency response).

Through both of these channels, preventive measures can mitigate risks – decreasing the (mean) number of deaths and the (mean) economic damages associated with environment-related natural disasters and industrial accidents. Nevertheless, the actual number of deaths and material damages suffered will remain as random variables (preventive measures can bring the mean down, although the variation will remain). Figure 21, which is based on European data, illustrates this point, with very large year-on-year variation in the number of floods, and associated deaths and damages.

Figure 21. Flood Disasters in Europe (1973 – 2002)



Source: Data from Hoyois and Guha-Sapir (2003); All values are scaled between 0 and 1.

To the extent that such (uncertain) events are due to anthropogenic factors, and to the extent that measures can be taken to reduce vulnerability in the affected areas, preventive activity still reduces their frequency and severity. In other words, humans can exercise a certain level of control over the stochastic process, in order to decrease the expected (mean) number of events and the associated costs. While preventive measures can mitigate the risks associated with environment-related hazards, prevention provides only imperfect control over the probabilistic process.

Society therefore faces a trade-off between prevention and remediation. Prevention requires *ex ante* expenditures – spending prior to an event – which may be politically unattractive because the benefits of the preventive measures are reaped only at some (unknown) point in the future. However, prevention can often be done at much lower cost than remediation. The available evidence suggests that preventive measures can deliver significant net benefits. For example, the World Bank and the US Geological Survey have estimated that the world-wide economic losses from natural disasters in the 1990s could have been reduced by $280 billion, if $40 billion had been invested in disaster preparedness, mitigation and prevention strategies (World Bank, 2004b).

An interesting question is, therefore, why the level of prevention seems to have been sub-optimal with respect to a number of environment-related disasters and accidents. Part of this may be attributable to the "external" nature of many of the costs of such impacts. If those responsible for environment-related hazards do not bear the cost of remediation and compensation, the level of prevention will be inefficient, because the environmental asset is treated as a "public good". This is certainly true of many types of natural disaster which are partly due to anthropogenic factors; it may also be true of industrial accidents.

The presence of important externalities may not be the only reason for inadequate prevention. Well-functioning markets internalise risk and uncertainty and allow investors to diversify risk (*e.g.* through insurance and financial derivatives). However, the insurability of large catastrophes may be compromised due to the ambiguity of risk[56] and the size of potential damages[57].

The role of public policy in this context is to provide appropriate incentives to encourage agents to mitigate risks associated with environment-related hazards. The choice of any policy alternative involves trade-offs between the potential levels of risk-reduction and the associated control costs. Optimal policy should provide incentives for polluters to adopt preventive measures or to exercise a level of precaution at which risk is reduced to its "optimum" level. At this point, the marginal cost of reducing the probability and impacts of hazards equals the marginal damage from avoided events.[58]

The costs of inaction with respect to such preventive strategies can be significant. According to Kunreuther and Michel-Kerjan (2007), "catastrophes" have had a more devastating impact on insurers over the past 15 years than at any other time in history. Data from Swiss Re and the Insurance Information Institute suggest that during the 1970s and 1980s, annual insured losses from natural disasters were in the 3-4 billion USD range. Since the latter part of the 1980s, the scale of insured losses from major natural disasters has exhibited a steep upward trend (Figure 22). The most recent data indicate that, of the 230 billion USD of economic damages inflicted by major *natural* catastrophes world-wide, a record amount of 83 billion USD was covered by insurance. On the other hand, the volume of insured losses from *human*-caused catastrophes (including industrial accidents) remained more-or-less constant over the entire period.

Environment-related industrial accidents and hazards

It is possible to get an idea of the scale of the problem related to environment-related industrial accidents from the Emergency Database of Disasters (EMDAT). A total of 1096 industrial accidents occurred worldwide between 1901 and 2006, affecting 4.2 million people, and inflicting economic damages of more than 20.3 billion USD. UNEP's *Programme on Awareness and Preparedness for Emergencies on a Local*

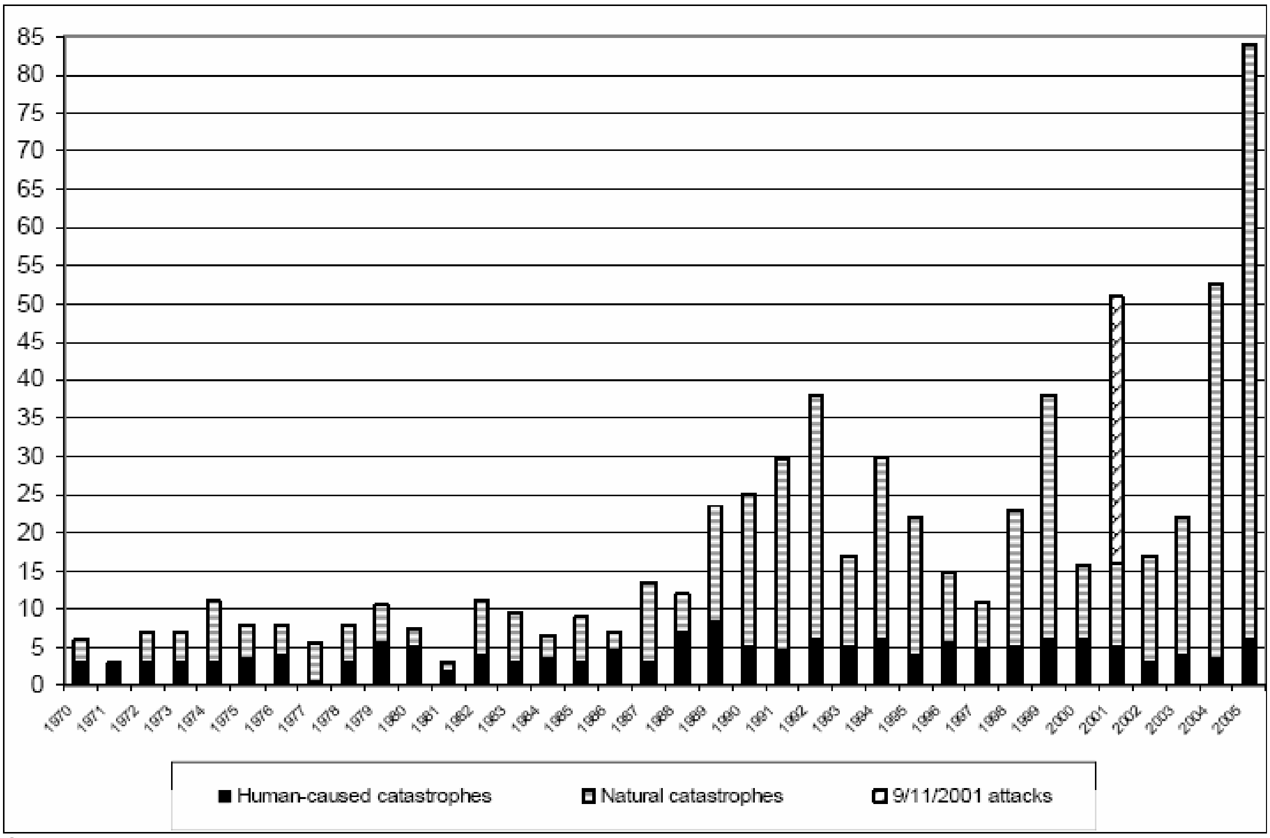
56 "Ambiguity of risk" refers to the difficulty of assigning probability to the risk. An insurer will be hesitant to provide coverage for a risk which s/he cannot understand (*i.e.* cannot specify the probability).

57 Insurers who cover risks from large-scale disasters (*e.g.* nuclear risks) may have to pay potentially large claims to policy-holders before they are to collect sufficient premiums to cover their costs. This problem of timing, and the desire to earn a positive expected profit, both make an insurer unlikely to insure catastrophic losses.

58 Policies which have the objective of eliminating risk entirely (*i.e.* reducing the number of events to zero) ignore the stochastic nature of events. In so doing, they impose excessive control costs on society.

Level keeps data on "technological disasters", which are defined to include transport-related accidents, nuclear accidents, storage facilities, fixed hazardous installations, ports and sea disasters, and tailings dam failures. Table 35 lists some of the most recent of those accidents involving hazardous substances, for which there have been at least 100 deaths.

Figure 22. Worldwide Evolution of Catastrophe-insured Losses, 1970-2005 (in Billion USD, Using 2005 Prices)



Source: Kunreuther and Michel-Kerjan (2007).

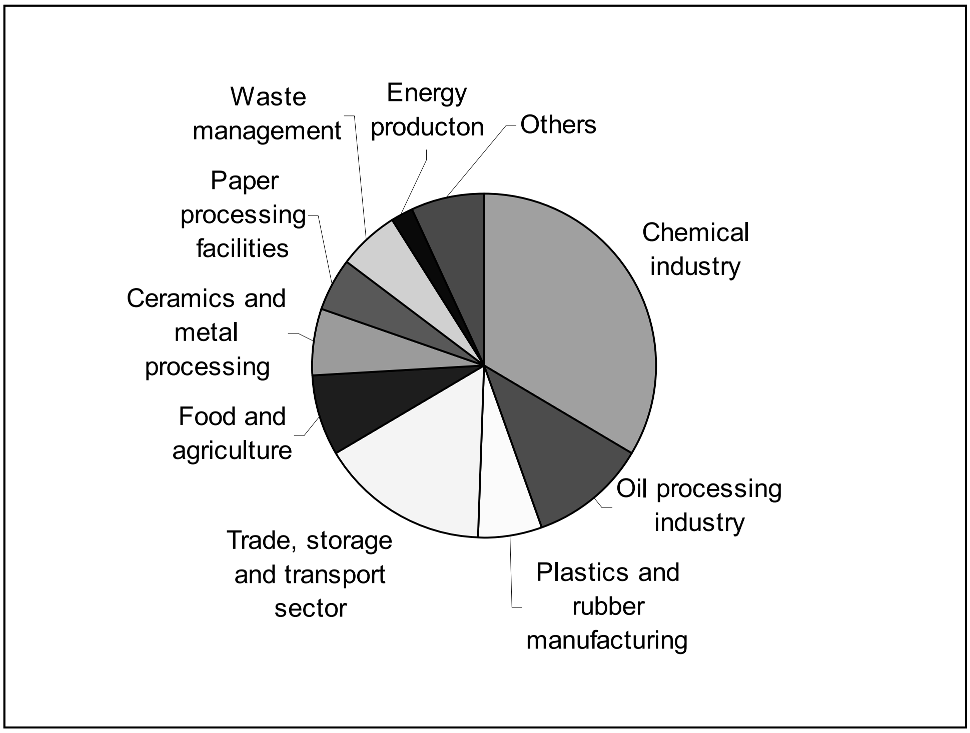
Columns 3 and 4 of Table 35 give an idea of the source of such incidents and the materials involved, and thus, the extent to which they can be linked to the objectives of environmental policy, rather than (say) work-place health and safety. In addition, European data on the sectors which are responsible for environment-related accidents is also available, since EU Member States must report all "severe" incidents to the registration centre of the Major Accident Hazard Bureau, which stores the data in the Major Accident Reporting System (MARS). The distribution of environmentally relevant incidents for 2005 is provided in Figure 23.

Table 35. Technological Disasters with at Least 100 Reported Deaths

Year	Location	Origin of accident	Products involved	Number Deaths	Injured
1984	India, Bhopal*	Leakage	Methyl isocyanate	2800	50 000
2002	Lagos, Nigeria	Ammunition Depot – Explosion and Firte	Explosives	2,000 (1,000 officially)	?
1989	USSR, Acha Ufa	Explosion pipeline	Gas	575	623
1975	India, Chasnala	Industry	-	431	..
1993	Columbia, Remeios	Release	Crude oil	430	
1983	Egypt, Nile River	Explosion (transport)	LPG	317	44
1979	USSR, Novosibirsk	Plant	Chemicals	300	..
2001	Lima, Peru	Fireworks spark – Fire		282	134
1993	Thailand, Bangkok	Fire in a toy factory	Plastics	240	547
1998	Cameroon, Yaoundi	Transport accident	Petroleum products	220	130
1978	Spain, San Carlos*	Road transport	Propylene	216	200
1992	Mexico, Guadalajara*	Explosion in the city sewers	Hydrocarbon oil, gas	>206	>1500
2001	Chehe, Guangxi, China	Mine accident (flood)	-	200	
2000	Payatas, Manila, Philippines	Landslides	Garbage dump	>196	
1991	Thailand, Bangkok	Transport accident	Dynamite, detonators	171	100
1988	UK, North Sea	Explosion, fire (platform)	Oil, gas	167	-
1982	Venezuela, Tacoa	Tank explosion	Fuel oil	>153	500
1990	India, Basti	Food poisoning	Sulplios	150	>150
1991	Italy, Livorno	Transport accident	Naphtha	141	
1996	China, Shaoyang	Explosion at a storage	Explosives	125	400
1980	Turkey, Danaciobasi	Use/application	Butane	107	..
1995	Korea, Taegu	Construction in the subway	LPG	101	140
1995	India, Madras	Transport accident	Fuel	~100	23
1995	Brazil, Boqueiro	Explosion at a store	Ammunition	100	
1991	Ethiopia, Addis Ababa	Explosion	Ammunition	100	200
1990	India, near Patna	Leakage, transport accident	Gas	100	100
1984	Romania	Factory	Chemicals	100	100
1978	Mexico, Xilatopec	Explosion (road transport)	Gas	100	200

Source: http://www.unepie.org/pc/apell/disasters/database/disastersdatabase.asp.

Figure 23. Distribution of Environmentally-relevant Incidents in the EU (2005)



Source: http://mahbsrv.jrc.it/mars/Default.html.

The main distinguishing features of environment-related industrial accidents – compared to environment-related natural disasters – are: (i) the greater possibility of exercising control over the probabilistic process; and (ii) the potential for the presence of asymmetric information:

- In the case of an oil spill, weather conditions (environmental uncertainty), vessel characteristics (technological uncertainty), as well as the level of skill and alertness of the personnel (human factor) all affect the likelihood of an accident. Even though the level of precaution significantly affects the probability of a potential accident, it is difficult for an outsider to observe the level of care actually being exercised by the responsible party.

- The problem arising from the presence of asymmetric information is that when one party (usually the polluter) has more information about the level of precaution being exercised than the other party (usually the regulatory authority), the party with more information has an incentive to "cheat" the party with less information.[59]

A policy may be imposed prior to the point at which an externality-generating activity takes place (*ex ante* instruments, such as design standards, restrictions on operations, corrective taxes, and other traditional regulatory approaches). Alternatively, policy incidence can be at the point at which damages from that

59 For more on "asymmetric information", see Arrow (1963), Vickrey (1961) and Akerlof (1970).

activity are realised (*ex post* instruments, such as nuisance laws and legal liability rules for environmental damages).[60]

The relative magnitude of costs (and their incidence) under *ex ante* and *ex post* approaches may differ substantially, depending on the nature of the risk (acute or chronic), the observability of actions (*i.e.* the associated monitoring and enforcement costs), the administrative complexity, or the number of parties involved and the time required to achieve an outcome (transaction costs). As a result, existing policies often combine elements of both systems. For example, the US Oil Pollution Act of 1990 (OPA-90) applies both: (i) design standards and operational guidelines (double-hull tanks and contingency planning), and (ii) liability rules and financial responsibility requirements (strict liability and mandatory insurance). The EU Directive on Maritime Transport mandates double-hull tanks similar to those of OPA-90.

Oil spills

The environmental impacts of oil spills can be significant. Even a relatively minor spill, depending on the timing and location of the spill, as well as the characteristics of the affected area, can cause significant harm to fragile marine and coastal ecosystems. Oil contamination may persist in the marine environment for many years after an oil spill. In exceptional cases, the effects may be measurable for decades after the event (see *e.g.* Kingston, 2002). The most visible environmental impacts of oil spills are those of acute mortality, persistence of toxic subsurface oil and chronic exposures. Even at sub-lethal levels, they may continue to affect wildlife and postpone the recovery of fragile coastal ecosystems over long periods. Recent scientific evidence has documented the chronic, delayed, and indirect long-term risks and impacts of oil spills (see *e.g.* Peterson *et al.,* 2003).

Figure 24 provides a representation of the kinds of costs arising out of an oil spill. The inner circle covers only remediation costs that are incurred by public authorities and/or the responsible parties. Values here are readily available, although even in this case the "cost" of volunteer efforts may be difficult to assess accurately. Moving outward from that circle, the values for losses which are reflected directly in terms of impacts such as lost opportunities for commercial fisheries are represented. In the next circle, other factors which do not find a "price" in existing markets are represented, such as amenity and recreation benefits which are not directly related to tourism. In the outer circle, all impacts are aggregated, including the costs associated with the non-use values of ecosystems. A spill in a remote environment may put rare ecosystems and species at risk. These ecosystems have a value, even if unexploited in any economic sense.

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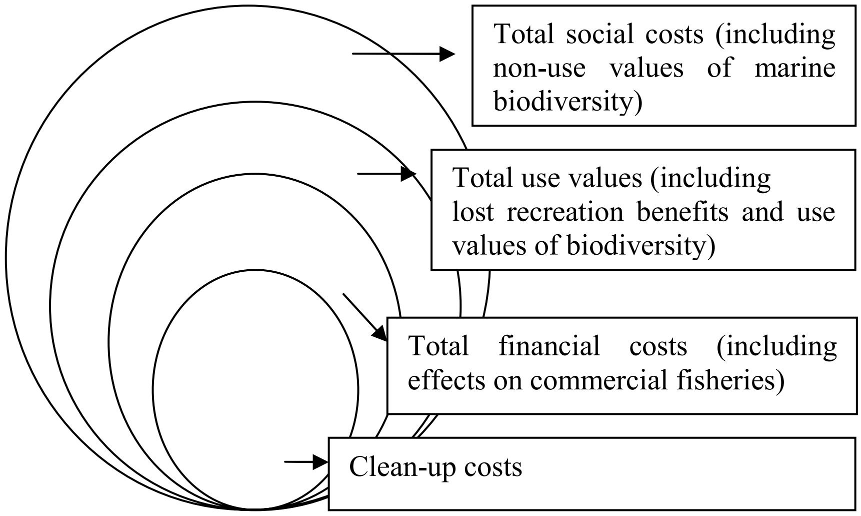
60 For example, a possible regulatory response to address hazards from oil spills is to set design standards (double-hull tanks, reliable navigation systems), to impose legal liability for damages, and to create favourable conditions in which insurance markets can develop.

Figure 24. Social Costs of Inaction with Respect to Oil Spills



World-wide, 1,716 of oil spills greater than 7 tonnes occurred between 1970 and 2005, involving a total of 5.6 million tonnes of oil (ITOPF, 2007). Table 36 lists the major spills which have occurred since 1967. The number of large spills has decreased significantly over the last three decades; the average number of large spills per year during the 1990s was less than one-third of that witnessed during the 1970s. Considering that consumption of oil has increased and the volumes of oil transported by vessels steadily increased world-wide during that period, the trend of declining oil spill incidents is noteworthy. Part of this decline can certainly be attributed to policy interventions (discussed below), as well as to exogenous technological improvements (*e.g.* improved weather forecasting) and endogenous market responses (*e.g.* the desire to avoid negative market impacts).

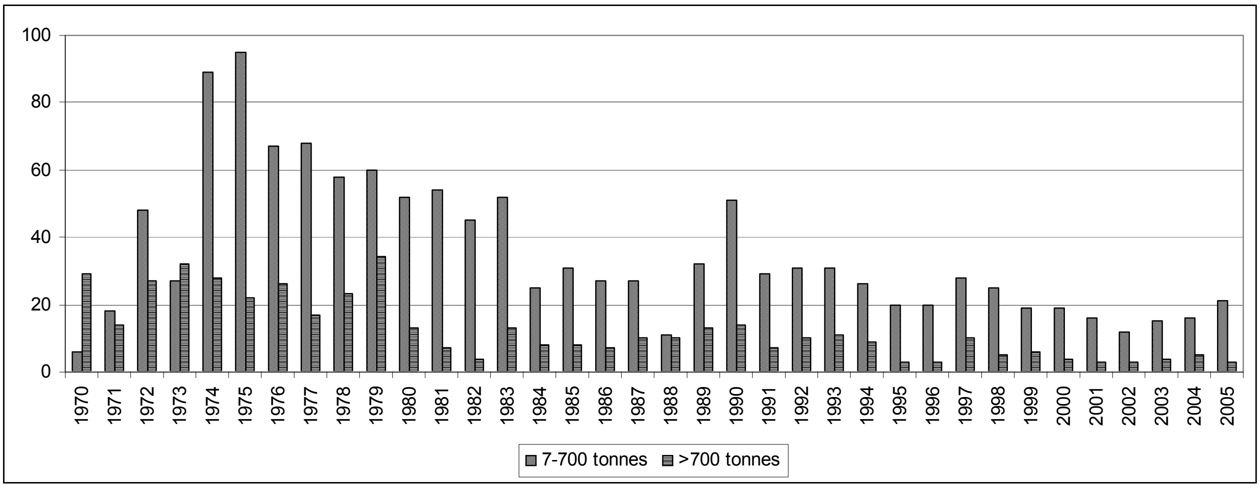
However, recent data suggest that the overall number of annual spill events may have ceased to decline. There has been virtually no decline in the mean number of spills during the decade since 1995 (Figure 25). If both consumption and volume of oil transported world-wide continue to grow, a higher percentage of transported oil will likely travel by vessel in the future. The risk of oil spills may thus increase. This raises the question of whether the existing international framework adequately addresses the risks from marine transport of hydrocarbons.

Table 36. Major oil Spills Since 1967

Ship name	Year	Location	Spill size (tonnes)
Atlantic Empress	1979	Off Tobago, West Indies	287,000
ABT Summer	1991	700 nautical miles off Angola	260,000
Castillo de Bellver	1983	Off Saldanha Bay, South Africa	252,000
Amoco Cadiz	1978	Off Brittany, France	223,000
Haven	1991	Genoa, Italy	144,000
Odyssey	1988	700 nautical miles off Nova Scotia, Canada	132,000
Torrey Canyon	1967	Scilly Isles, UK	119,000
Sea Star	1972	Gulf of Oman	115,000
Irenes Serenade	1980	Navarino Bay, Greece	100,000
Urquiola	1976	La Coruna, Spain	100,000
Hawaiian Patriot	1977	300 nautical miles off Honolulu	95,000
Independenta	1979	Bosphorus, Turkey	95,000
Jakob Maersk	1975	Oporto, Portugal	88,000
Braer	1993	Shetland Islands, UK	85,000
Khark 5	1989	120 nautical miles off Atlantic coast of Morocco	80,000
Aegean Sea	1992	La Coruna, Spain	74,000
Sea Empress	1996	Milford Haven, UK	72,000
Katina P	1992	Off Maputo, Mozambique	72,000
Nova	1985	Off Kharg Island, Gulf of Iran	70,000
Prestige	2002	Off the Spanish coast	63,000
Exxon Valdez	1989	Prince William Sound, Alaska, USA	37,000

Source: ITOPF (2007).

Figure 25. Number of Oil Spills Over 7 Tonnes, 1970 – 2005, World-wide



Source: ITOPF (2007).

These large-scale incidents may be devastating for local communities. Coastal economies are typically resource-based, and therefore heavily dependent on the status of their marine and coastal resources. Large-scale damage to that resource base may seriously endanger local industries (fisheries, fish-processing, and tourism sectors), which typically form the backbone of coastal economies (see *e.g.* Garza-Gil *et al.,* 2006).

Compensation for the damages caused by oil pollution is currently based on two international conventions: the Civil Liability Convention (CLC) and the Fund Convention, adopted under the auspices of the International Maritime Organisation. Those who suffer losses as a result of an oil spill therefore have relatively easy access to compensation, without the necessity of complex litigation.[61] Five types of damages are admissible for compensation claims under the IOPC: property damage; cleanup costs; economic losses of the fisheries sector; economic losses of the tourism sector; and environmental restoration costs.

The CLC provides a first tier of compensation, which is paid by the owner of a ship which causes the pollution damage. The ship-owner has strict liability for damages. However, his/her financial liability is limited to an amount determined by the tonnage of the ship. This amount is guaranteed by the shipowner's liability insurer. The International Oil Pollution Compensation (IOPC) funds provide supplementary compensation to victims who are unable to recover all of their losses from the ship-owner's limit of liability under CLC. IOPC funds are financed by industry's contributions, on the basis of volumes of oil received. The amount of the contribution depends on the quantity of oil received (IOPC, 2007). Even though IOPC funds provide an additional tier of compensation, the total amount of funds available for a given incident is limited. Currently, the total amount of compensation available is 1,079 million USD[62] (this is cumulative for all CLC and IOPC funds).

The case of the *Exxon Valdez* is interesting, because it provides lessons on the relative costs of prevention and remediation, as well as on the incentives faced by the responsible party. In 1989, the *Exxon Valdez* ran aground and spilled 37,000 tons of oil into Prince William Sound in Alaska (US). The spill eventually spread over 10,000 square miles of water and 1,000 miles of shoreline. A valuation study by Carson *et al.* (1992) assessed the damage at USD 2.8 billion.[63]

The actual damage award deviated significantly from this estimate. The settlement accepted in 1991 by the US District Court was USD 1.025 billion (Harrison, 2006). This included a criminal fine of 150 million, 125 million of which was forgiven in recognition of Exxon's cleanup efforts. Exxon agreed to pay USD 100 million to the trustees as criminal restitution for the injuries caused to fish, wildlife and land. Exxon also agreed to pay USD 900 million to the trustees over a 10-year period, in order to settle the civil actions. The settlement allowed for a possible additional USD 100 million, if resource damage occurred that could not have been anticipated at the time of the settlement (Harrison, 2006).

However, the costs incurred by Exxon have been even greater than this. In addition to the settlement of the state and federal lawsuits, Exxon paid USD 2.2 billion and USD 300 million for lost wages to 11,000 fishermen and business firms. The cost to the fisheries of south-central Alaska has been estimated at 108.1 million, the largest component being a 65.4 million reduction in the pink salmon fishery in the first year following the accident (Cohen, 1995). In 1994, an Alaska jury awarded an additional USD 5.3 billion in punitive and compensatory damages to those harmed by the Exxon Valdez oil spill. Exxon's appeal was rejected by an Alaska Appeals Court in March 2000 (Talley, Jin and Kite-Powell, 2005).[64]

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61 This is particularly important in the absence of the possibility of class-action suits.

62 The amount is specified in Special Drawing Rights (SDR); the 1,079 USD is based on USD/SDR exchange rate of 1 SDR = 1.44 USD (in March 2006).

63 An abridged version of the full report was published as Carson *et al.* (2003).

64 Punitive damages are awarded only exceptionally in order to deter the defendant and similar persons from pursuing a course of action, such as one which damaged the plaintiff. Punitive damages are thus separate (and in excess) of the compensatory damages awarded to a plaintiff. It has been argued that punitive liability increases deterrence, because it provides incentives for potential defendants to reduce risk or to

Thus, the total cost of the incident to Exxon itself (Table 37), was well in excess of some estimates of the damages. The difference is largely explained by the "punitive" character of the 1994 damage award. It is unclear to what extent this award reflects losses in amenity and ecosystem services. While these values (and others) are certainly included in the damage assessments by Carson *et al.* (1992), this estimate should be considered independently, in order to avoid double-counting.

Table 37. Costs of Exxon Valdez Oil Spill to Exxon

	Estimate (million USD)	
Cleanup	2,200	
State and federal lawsuits	1,000	
Compensation for lost wages to fishermen and businesses	300	
of which - pink salmon fishery		65.4
- other fisheries		108.1
- other industry losses		191.9
Additional punitive and compensatory damages	5,300	

Source: Carson *et al.* (1992); Cohen (1995); Talley, Jin and Kite-Powell (2005); Harrison (2006).

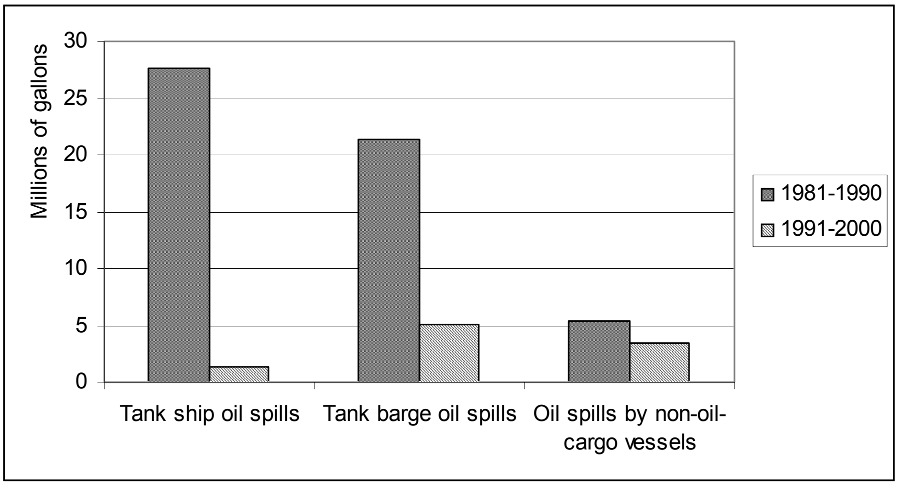
The Exxon Valdez spill led to the Oil Pollution Act of 1990 (OPA-90), which strengthened accountability for vessel oil spills in US waters. It limited the strict liability of the vessel owner, but specified unlimited liability if the vessel owner (or other liable party) was found guilty of gross negligence, or in violation of laws. In addition, OPA-90 requires vessels to carry certificates of financial responsibility, proving that the owners have funds (or insurance) sufficient to cover the maximum liability limits allowed under the law. OPA-90 requires the use of double hulls for tankers by 2015, to reduce the likelihood or severity of damage to a vessel's tanks. It also requires interim structural and operational measures, aimed at reducing the outflow of oil, in the event of an oil spill (Talley, Jin and Kite-Powell, 2005).

The total number and volume of oil spills in US waters from tank ships and barges declined considerably after enactment of OPA-90 and its financial responsibility regulations (Figure 26). While it is difficult to determine precisely how much of this decline can be attributed to incentives provided by OPA-90 and by the signals provided by the Exxon Valdez settlement, the deterrent effect was likely to have been considerable (OCIMF, 2003).

The situation in Europe is quite different, and three spills can be used to illustrate the differences. The social costs of the *Amoco Cadiz* oil spill were estimated at EUR 447 million (Bonnieux and Rainelli, 1991) (Table 38). This includes EUR 230.5 million in cleanup and restoration costs, EUR 50.6 million of marine resources losses (fish and shellfish industry), EUR 85.7 million in tourist industry losses, EUR 51.7 million in recreation and amenity losses, and EUR 80.2 million in ecological losses.

avoid being found punitively liable. According to Polinsky and Shavell (1998), the goals of punitive damages are deterrence and punishment. The deterrence rationale for punitive damages is two-fold: punitive damages should be awarded if, and only if, an injurer has a significant chance of escaping liability for the harm caused; and punitive damages may be needed to deprive individuals of the socially illicit gains that they obtain from malicious acts.

Figure 26. Volume of Oil Spills, US



Source: US Coast Guard (2002).

Table 38. Social Costs of the Amoco Cadiz Oil Spill

	Estimate (million 2003 euros)		%
Cleanup and restoration	230.5		46.2%
of which - at sea		27.0	
- at land		111.4	
- substructures restoration		92.1	
Marine resources losses	50.6		10.1%
of which - fishing industry		22.6	
- shellfish breeding		28.0	
Tourism	85.7		17.2%
of which - direct losses (value added)		55.4	
- indirect losses to other sectors		30.3	
Recreation and amenity losses	51.7		10.4%
Ecological losses[65*]	80.2		16.1%
Total (estimated) social costs	447.0		100.0%

Source: Bonnieux and Rainelli (1991).

On December 16, 1999, the tanker *Erika* ruptured off the south coast of Brittany (France). Over the next six months, the spill spread along 400 km of coastline. Total damages were estimated at EUR 914 million (as reported in Bonnieux and Rainelli, 2003), including EUR 124 million for land-based cleanup and restoration costs, EUR 52-73 million in marine resources losses, and EUR 400-500 million in lost receipts to the tourism industry (Table 39). The amenity damages were also evaluated by Bonnieux and Rainelli (2003) at EUR 98.3 million. No estimate of the value of lost ecosystem services (ecological losses) is available.

65 Ecological losses were assessed on the basis of the value of commercial species corresponding to the loss of the non-commercial biomass.

Table 39. Social Costs of Erika Oil Spill

	Estimate (million 2000 EUR)	
Cleanup and restoration[66]		
of which – at sea		n.a.
- at land		124
- other		n.a.
Marine resources	52-73	
Tourist industry	400-500	
Total estimated costs	914	
Recreation and amenity losses	98.3	
Ecological losses	n.a.	
Total social costs	*n.a.*	

Source: Data from Bonnieux and Rainelli (2003).

The social costs caused by the *Prestige* oil spill off the Galician coast (Spain) in 2002 were evaluated by Loureiro *et al.* (2006). They estimated that the short-term losses in all affected economic sectors, as well as the cleanup and recovery costs, and all environmental losses accountable at the time, generated a lower bound estimate of EUR 770.6 million (2001 prices), excluding all other financial and future costs (Table 40). According to this estimate, the total short-term (2002-2004) costs of the oil spill amounted to EUR 567 million in Galicia alone, representing 1.57% of its annual GDP. The economic magnitude of the catastrophe was therefore very significant.

Table 40. Social Costs of the Prestige Oil Spill for All Affected Areas

	Estimate (million EUR)	
Cleanup and restoration costs	509.4	
Of which – Cleaning and recovery costs minus residual value of investment		228.0
- Extraction of fuel remaining inside tanker		100.0
- Recycling of residuals		32.0
- Expenditures by local communities and governments		123.5
- Volunteers		4.0
- Losses in other goods		2.6
- Advertising and promotion		19.0
- IGAPE support to the mussel sector		0.3
Marine resources (2002-04)	152.3	
Of which – Fisheries sector		112.7
- Mussel sector		12.8
- Canning and fish processing sector		26.8
Tourism sector (2002-03)	110.6	
Environmental losses: Birds and mammals	25.1	
Total estimated costs	*770.6*	
Recreation and amenity losses	n.a.	
Ecological losses (other than above)	n.a.	
Total social costs	*n.a.*	
Other expenditures		
- Transfers to fishermen and shellfish pickers while fishing and shellfish bans were in place		134.3
- Other compensations to private parties		94.0

Source: Adapted from Loureiro *et al.* (2006).

Cost estimates of similar order of magnitude were also estimated by Garza-Gil *et al.* (2006), who focused on the economic damages inflicted in the Region of Galicia (Table 41). Both Loureiro *et al.* (2006) and Garza-Gil *et al.* (2006) suggested that the economic losses arising from the *Prestige* spill exceed those that

66 Final cost estimates are not yet available. Claims are still being processed.

can be indemnified under the International Oil Pollution Compensation (IOPC) system. The magnitude of the latter could exceed by five times the applicable limit of compensation in the *Prestige* case.[67]

Table 41. Social Costs of the Prestige Oil Spill for the Galicia Region (Spain)

	Estimate (million EUR)
Cleaning and restoration	559.0
Of which - at sea	180.0
- at land	315.0
- remaining oil extraction	60.0
Coastal fisheries and aquaculture (2003 only)	64.9
Tourism (2003 only)	133.8
Total estimated costs	*761.7*
Recreation and amenities	n.a.
Ecological losses	n.a.
Total social cost	n.a.

Source: Adapted from Garza-Gil *et al.* (2006)

In the case of *Erika*, a decision was issued by a French court on January 16, 2008 in a case involving 101 plaintiffs and 15 accused parties. The court found TOTAL S.A. and three other parties (the owner and the manager of the Erika tanker, and RINA - the Italian classification company) guilty of negligence and ordered them to each pay a share of the 900,000 EUR fine and 192 million EUR of damages to some of the 101 plaintiffs in the case. While the victims have collectively claimed up to one billion EUR in damages, some of the 101 claims were not admissible in court, according to the ruling.[68] In addition, the damage award primarily covers clean-up costs and costs to commercial fisheries and the tourism industry. While the principle of ecological losses has been recognized, the costs are not really covered, except for a very small amount (1,3 million EUR) awarded to the Department of Morbihan for damages to "sensitive coastal areas" and to the LPO (League for Protection of Birds) for damage to bird populations.

According to IOPC, a total of EUR 171.5 million is available for compensation payments for the *Prestige* incident (EUR 22.8 million from the ship-owner's liability insurer and EUR 148.7 million from the IOPC Fund), against 1315 claims (from Spain, France, and Portugal), totalling EUR 866.8 million. In order to provide all claimants with equal treatment, the compensation payments have been limited to 15% of the loss or damage actually suffered by each individual claimant (the limit was later increased to 30%) (IOPC, 2007).

For comparison, Garza-Gil *et al.* (2006) have estimated the cleaning and restoration costs actually incurred as a percentage of total damages for three spills (*Amoco Cadiz, Exxon Valdez,* and *Prestige)*. Exxon paid for all mitigation costs in the *Exxon Valdez* spill, and provided funding to cover restoration costs, equivalent to 100% of assessed damages. A much smaller proportion of the clean-up and restoration costs have been recovered in the *Amoco Cadiz* (50%) and *Prestige* (15%) cases (Table 42). In its recent court decision the *Tribunal de Grande Instance* awarded just over 160 million EUR in compensation for clean-up costs associated with the *Erika* spill, principally to the French State.

67 The applicable limit of compensation prior to the 2003 amendment was $180 million.

68 TGI (2008) Jugement, 16 Janvier 2008, #9934895010. Tribunal de Grande Instance de Paris, France. pdf version of the judgement available at: http://www.proces-erika.org/articles/article/article/video-proces-erika-peine-maximum-pour-total/index.html?tx_ttnews%5BbackPid%5D=4938&cHash=ca7bcfa4ab

Table 42. Cleaning and Restoration Costs in Selected Oil Spills

	Volume spilled	Estimated cost	Cost per ton	Compensation as % of cleaning and restoration costs
Amoco Cadiz (1978)	223,000 tonnes	134 million EUR	650 EUR	50%
Exxon Valdez (1989)	35,000 tonnes	3,100 million USD	70,454 USD	> 100%
Prestige (2002)	77,000 tonnes	559 million EUR	10,666 EUR	15%

Source: Garza-Gil *et al.* (2006).

The sinking of the *Prestige* in 2002 accelerated the timetable for new standards in Europe, similar to those of OPA-90 (Talley, Jin and Kite-Powell, 2005). These have been implemented (or will be implemented) according to the following timetable:

- Erika I package (2001/105/EC)
  - Accelerated phasing-out of single-hull tankers;
  - Reinforced inspection regime.

- Erika II package (2002/59/EC)
  - Established a European Maritime Safety Agency (EMSA);
  - Established EU monitoring, control, and info system for maritime traffic.

- Erika III package (proposal)
  - Improve accident and pollution prevention (inspections, traffic monitoring);
  - Harmonise framework for investigation of accidents.

Much of the inspiration for the relevant EU Directives comes from the OPA. Thus, the regulatory standards are similar. In addition, while the liability regimes were quite different in the past, they are now becoming more similar. Nonetheless, important differences remain. In the case of the US, the OPA sets the framework for liability and compensation; in Europe, liability is covered by the international Convention on the Civil Liability (CLC) for Oil Pollution Damage. Both regimes apply a principle of strict liability. However, under the OPA, liability is unlimited in the case of negligence. Moreover, there is potential for punitive damages.

Under the CLC, liability is always capped. The compensation fund was increased in principle in 2003 from USD 180 million to a maximum of USD 1 billion available per spill. More significantly, the only costs that can be compensated under the fund are those which relate to clean-up and restoration, losses in the fisheries and the seafood sector, and coastal tourism. Lost recreation opportunities which are not marketed and cultural, existence and heritage values cannot be compensated (Garza-Gil *et al.*, 2006).[69]

The estimated costs of the past oil spills presented in this Section illustrate that the damages inflicted by these incidents have been considerable. It is likely that the costs of having avoided the oil spills would have been less than the benefits of having behaved preventively. From the perspective of "costs of inaction", it is important to provide incentives for responsible parties to exercise adequate level of precaution. The damage estimates presented here give an indication of the orders of magnitude associated with accidents in maritime transport of hydrocarbons.

[69] Given the post-2003 liability regime, if incidents similar to those of the *Erika* and *Prestige* spills happened in the future, the new liability limits would be sufficient to cover the applicable damage claims. However, the costs of the *Exxon Valdez* spill would be well in excess of the new liability limits. In addition, the new liability limits still only apply to three types of costs – namely cleanup costs, restoration costs, and losses to fisheries and tourism industries – leaving other losses potentially uncompensated.

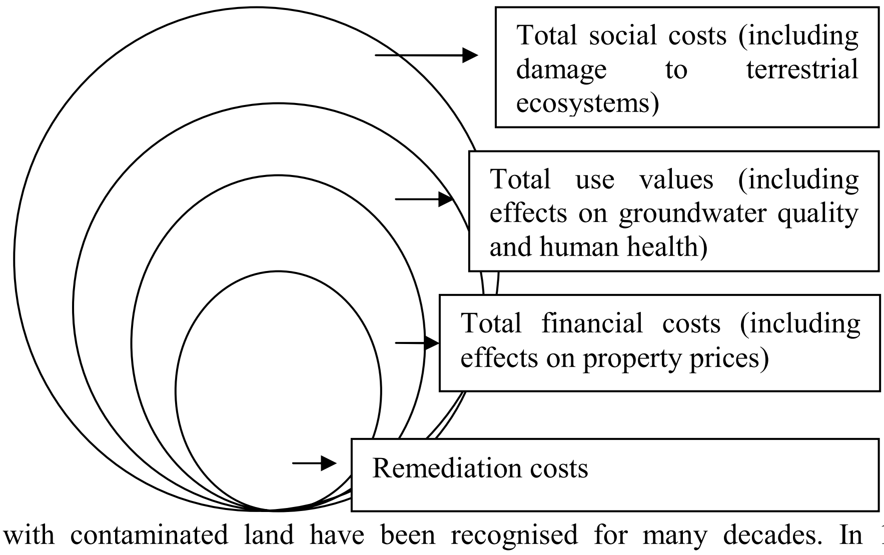
Contaminated land

Like oil spills, hazardous waste generation may pose a significant threat to both human health and ecosystems. High-risk waste contains significant concentrations of substances that are highly toxic, mobile, persistent and/or bio-accumulative. This has resulted in a large amount of "contaminated land" in OECD countries which is not suitable for development, and which may pose significant health risks in the absence of remediation. Although the costs of avoiding such a situation may have been very low in hindsight, the costs of remediation today (and likely, in the future) are often very high.

While the adverse effects of contamination may not be immediately apparent, the long-term impacts (via continued leakage of hazardous substances into surface and ground water, and possibly entering the food chain) may be detrimental to environmental quality, and may be accompanied by human-health risks. Several health effects of primary concern may affect populations exposed to hazardous chemicals (Grisham, 1986), including carcinogenesis, genetic defects, reproductive abnormalities, alterations of immunobiological homeostasis, central nervous system, and congenital anomalies. Exposure can lead to a reduction of life expectancy and possibly to reduced quality of life.

Figure 27 illustrates the kinds of costs arising from a contaminated site. *Ex post* remediation costs are reflected in the innermost circle. Moving out from that circle, the effects on property prices represented, and are generally obtainable.[70] However, lost development opportunities can be difficult to value. The next circle represents use values which can be difficult to value due to significant uncertainty – *i.e.* effects on future groundwater availability or health effects with long latency periods and uncertain epidemiological evidence. In the outer circle, all impacts are aggregated, including the costs associated with damages to the non-use values of terrestrial ecosystems.

Figure 27. Social Costs of Inaction With Respect to Contaminated Land



The problems associated with contaminated land have been recognised for many decades. In 1980, CERCLA was introduced in the US, to identify high-priority sites and to initiate remedial cleanup. Sites identified for cleanup are formally listed on the National Priorities List (NPL). Remedial operations are financed out of "Hazardous Substances Trust Fund" (commonly referred to as "Superfund").

70 In principle, these should include some of the values included in the outer circles, but this depends on the existence of full information and efficient markets.

The law imposes liability on firms, municipalities, and other "potentially responsible parties" (PRPs) for cleaning up contaminated sites selected by the US Environmental Protection Agency (USEPA). The set of PRPs includes nearly anyone involved with the waste at some point between generation and final disposal. Courts have interpreted CERCLA as imposing strict, retroactive, and joint and several liability, meaning that a single firm may be held liable for all remediation costs at a site, even if the firm contributed only a small proportion of the hazardous materials, and acted in accordance with established legal practice of the time. Broad interpretation of liability coupled with broad definition of PRPs encourage private funding of cleanup and reduce the amount of cleanup that must be financed (without reimbursement) out of the Superfund (*i.e.* from public funds) (Segerson, 1997).

Total cleanup costs have been estimated at USD 30 billion for the period between 1981 and 2000, with USD 43 million average cost per site. Future (undiscounted) Superfund cleanup costs have been estimated at USD 100-300 billion over 30 years (Russell, Colglazier and English, 1991, cited in Garber and Hammitt, 1998). The costs of the program to PRPs are thus expected to be substantial. The chemical industry is expected to bear 25% of total Superfund costs, suggesting industry costs in the order of USD 1 billion per year (Probst *et al.,* 1995).

Much of the cost can be attributed to transaction costs.[71] For example, Acton and Dixon (1992) reported that transaction costs were 19% of outlays for five very large industrial firms at 49 sites on the NPL between 1984 and 1989. These firms had annual revenues in excess of USD 20 billion. In another study, Dixon, Drezner and Hammitt (1993) presented information on the expenditures of 108 firms with annual revenues less than USD 20 billion between 1981 and 1991 at 18 sites on the NPL. They found that individual firm expenditures and transaction-cost shares[72] vary enormously by firm size: transaction-cost share averaged 60% for firms with annual revenues less than USD 100 million, 15% for firms with annual revenues between USD 100 million and 1 billion, and 19% for firms with annual revenues between USD 1 and 20 billion. The overall transaction cost share was estimated at 32% of total expenditures by PRPs through 1991 at all sites on the NPL (Dixon, Drezner and Hammitt, 1993).

The same study also estimated that transaction costs for insurers were even higher – 88% of outlays between 1986 and 1989. These estimates imply that 36% of the approximately USD 11.3 billion spent by the private sector at Superfund sites through 1991 went to pay transaction costs, rather than to support cleanup. Many of these transaction costs relate to insurance claims. According to Dixon, Drezner and Hammitt (1993), among the PRPs with expenditures on coverage disputes over USD 1000, only 12% of firms received reimbursement. The reimbursements were over six times the firm expenditures on coverage disputes. Overall, insurers reimbursed PRPs for approximately 8% of their cleanup expenditures.

Legal liability for environmental damages may also decrease the value of a company's stock, in response to increased investors' expectations of future costs. However, greater exposure to environmental liability may impose financial risks on investors and therefore increase firms' costs of capital. This is because investors require higher expected returns on securities they perceive to be riskier. Although the liability for past waste disposal practices may appear unavoidable (sunk), it could affect future business decisions by raising a firm's cost of capital. Increases in the costs of capital represent social costs of bearing financial risk. Garber and Hammitt (1998) found that social costs of bearing financial risks associated with Superfund liability are substantial. The costs ranged from about USD 200 to 350 million annually, using the 1981-1992 estimates and USD 685 to 820 million, using the 1988-1992 estimates. These costs are a substantial fraction of total chemical industry cleanup costs.

71 Transaction costs equal expenditures incurred in assigning liability among parties involved at a site.

72 Transaction-cost share = the ratio of transaction costs to the sum of transaction costs and investigation and remediation costs.

Messer *et al.* (2006) analysed the long-term impacts of delayed cleanup on property values in communities neighbouring prominent Superfund sites (34,000 homes near sites in three metropolitan areas, for up to a 30-year period). They found that, when cleanup is delayed for 10, 15, or even 20 years, the discounted present value of the cleanup is mostly lost. A possible explanation for these property value losses is that the sites are stigmatized, and that homes in the surrounding communities are shunned. This suggests that expedited cleanup and minimizing the number of stigmatizing events would reduce these losses.

Beyond the effects on property prices, the problem of contaminated lands (and efforts to address the problem through CERCLA) may have more far-reaching effects on land markets. In particular, it has been argued that the retroactive statutes (a party purchasing a piece of property may inherit potential liability) may discourage property sales and redevelopment of brown-fields. In addition, the lender liability statutes (broad definition of PRPs) may reduce the availability of funds both for investment and real estate purchases (Segerson, 1997).

Hamilton and Viscusi (1999) carried out a comprehensive analysis of the relative levels of risk-reduction and cleanup costs, using a sample of 150 Superfund sites in 1991-92, for which chemical analysis and risk assessment data were available. They calculated that the sites would yield 731 cases of cancer in the next 30 years, in the absence of any cleanup. However, they also showed that there is less than a 1% probability that anyone will ever be exposed to the concentrations that USEPA routinely assumes in assessing cancer risks. They found that, at the majority of sites, the expected number of cancers averted by remediation is less than 0.1 cases per site, and that the cost per cancer case averted is over USD 100 million. That is, of course, far in excess of the value of approximately USD 5 million placed by EPA on the value of a life saved. In another study, Viscusi and Hamilton (1999) found that virtually all of the expected cancer cases that are reduced – over 99% – are prevented by the first 5% of the expenditures.

Gayer, Hamilton and Viscusi (2000) estimated the value that residents themselves actually place on avoiding cancer risks from hazardous waste sites, based on a sample of 16,928 properties near 7 Superfund sites. They estimate an upper-bound benefit of cleanup USD 9.1-10.1 million for a reduction in the mean level of cancer risk. For comparison, the total present value cost of the EPA's remediation plans for the selected sites is USD 56.8 million. Gayer *et al.* concluded that, had the EPA undertaken only institutional controls for the remediation, the total present value cost would be USD 5.4 million -- a figure more consistent with values implied by residents' private valuation of cancer risk reduction. In contrast, Kiel and Zabel (2001) found that the benefits from cleaning up two Superfund sites in Woburn, Massachusetts are in the range of USD 72 - 92 million (1992 dollars) which, they concluded, is likely to be greater than the present value of the estimated costs of cleaning up these sites (thereby yielding positive net social benefits).

The costs of CERCLA, although considerable, may still be less than the benefits. However, it is also likely that the costs of having avoided the initial contamination would have been less than the benefits of having behaved preventively. From the perspective of "costs of inaction", it is important to ensure that land does not become contaminated in the first place, via inappropriate disposal of hazardous wastes. In this respect, CERCLA does provide very strong incentives to exercise caution.

While the policy response has been somewhat different, the magnitude of the costs of inaction with respect to contaminated land is very similar in Europe. It has been estimated that there are over 400,000 contaminated sites in the EIONET countries[73] [74] [75] (EEA CSI, 2006), and this number is expected

[73] Austria, Belgium, Bulgaria, Cyprus, Czech Republic, Denmark, Estonia, Finland, France, Germany, Greece, Hungary, Iceland, Ireland, Italy, Latvia, Liechtenstein, Lithuania, Luxembourg, Malta, Netherlands, Norway, Poland, Portugal, Romania, Slovakia, Slovenia, Spain, Sweden, Switzerland, Turkey, and the UK.

to grow by over 40% by 2025. Heavy metals and mineral oils are the most frequent sources of soil contamination. For groundwater, mineral oils and chlorinated hydrocarbons are common problems. Remediation efforts are underway, but it has been estimated that less than 60,000 sites have been "cleaned up" in the last thirty years (EEA CSI, 2006).

The costs of remediation are considerable. Annual expenditures represent approximately 0.05-0.1% of GDP in the countries for which data is available, although there are a small number of European countries for which the costs are much higher (EEA CSI, 2005) (Figure 28.) More significantly, this only represents approximately 2.5% of the total estimated remediation costs – *i.e.* the undiscounted value of costs of remediation is between 2% and 4% of a single year's GDP.

In principle, the private sector should bear responsibility for most of these costs (as well for the ecological and other damages). A review of national legislation which is relevant for the management of contaminated sites in Western Europe, EEA (2000) found that, even though most countries formally ascribe to the "polluter pays principle" (*e.g.* Part IIA of the UK's Environmental Protection Act), the actual practice is somewhat different. This gap in implementation is due in part to difficulties in identifying responsible parties, and to cases in which the relevant parties are no longer operating.

As a consequence, the public sector (taxpayers) can bear a share of the financial burden of remediation. At the EU level, the rehabilitation of industrial sites is funded through Structural Funds, with a budget of 2.25 billion in the EU-25 (EEA CSI, 2006). On average, 35% of total remediation costs are borne by the public sector (in some countries, this figure is 100%) (Figure 29).

While there is no over-arching EU policy for contaminated sites, the incentives of potentially responsible parties are affected by the European Union's Directive on Registration, Evaluation, and Authorisation of Chemical Substances (REACH) (Directive 2006/121/CE). REACH replaced the previous chemicals regulatory system, with the intention to bring within the scope of the authorisation system all substances of high concern, to make data publicly available, and to encourage innovation to develop alternative safer chemicals. REACH attempts to overcome shortcomings in the previous chemicals policy (slow and ineffective testing). In particular, REACH: (i) shifts the status of a chemical from "presumed safe" to "presumed unsafe"; and (ii) shifts the burden of proof from consumers to producers.

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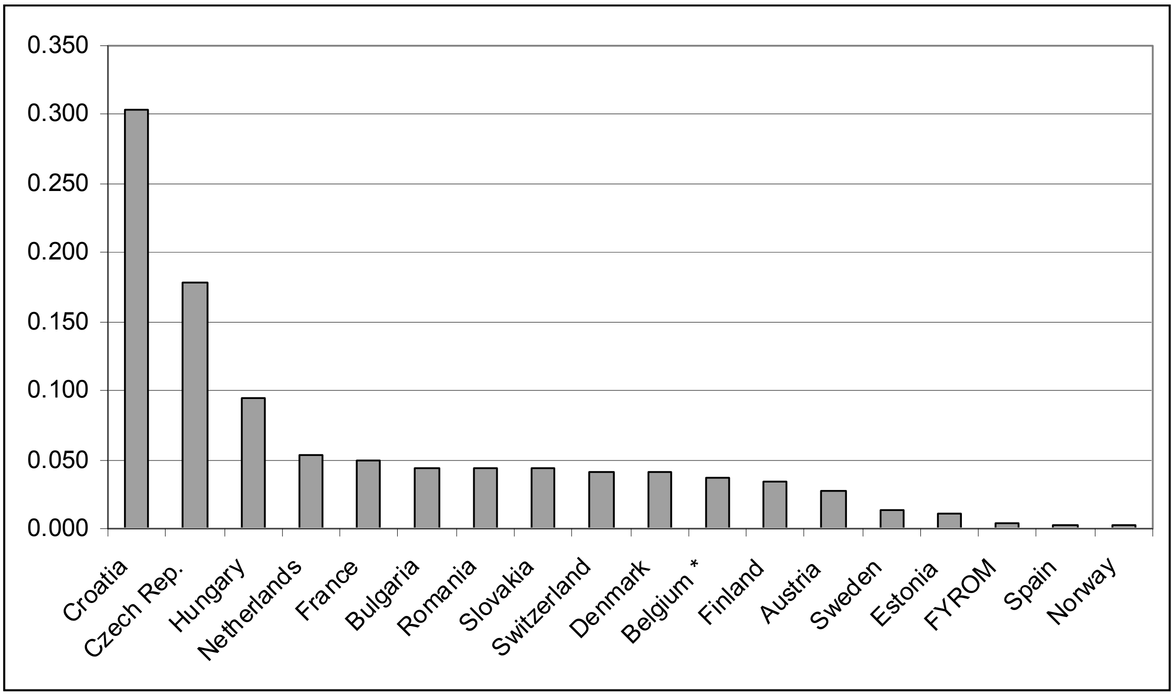
[74] Footnote by Turkey: The information in this document with reference to « Cyprus » relates to the southern part of the Island. There is no single authority representing both Turkish and Greek Cypriot people on the Island. Turkey recognises the Turkish Republic of Northern Cyprus (TRNC). Until a lasting and equitable solution is found within the context of United Nations, Turkey shall preserve its position concerning the "Cyprus issue".

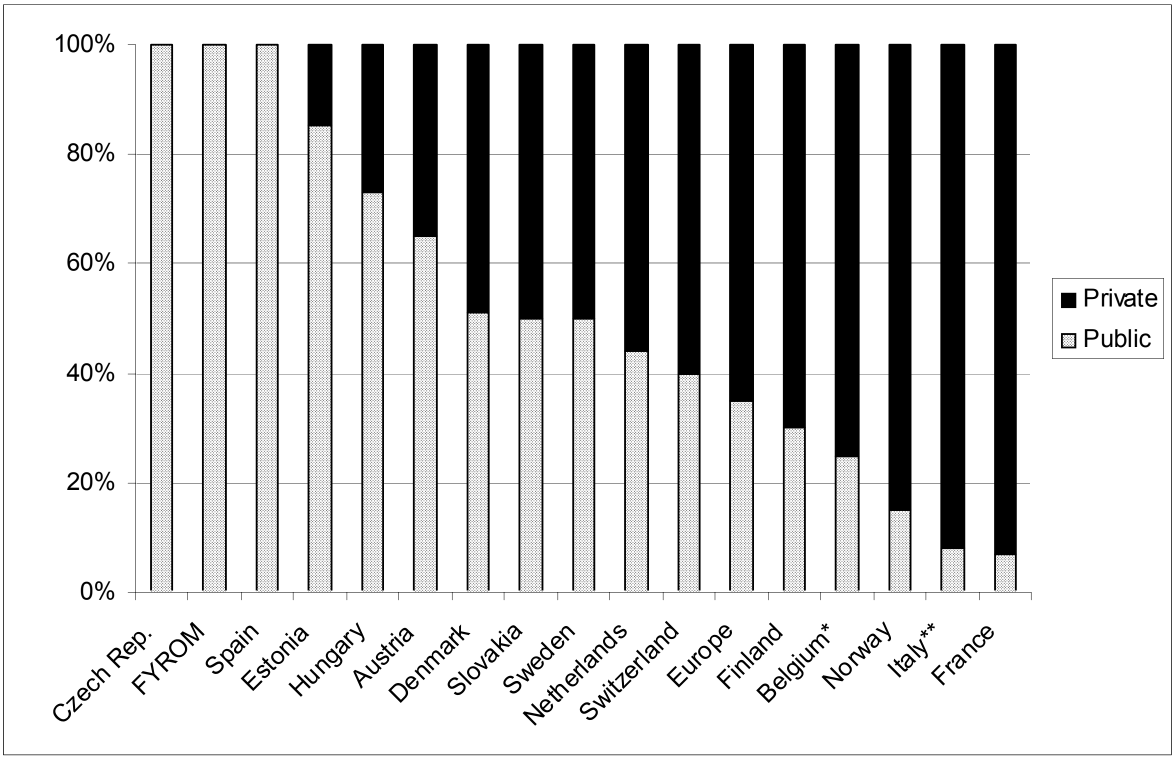
[75] Footnote by all the European Union Member States of the OECD and the European Commission: The Republic of Cyprus is recognised by all members of the United Nations with the exception of Turkey. The information in this document relates to the area under the effective control of the Government of the Republic of Cyprus.

Figure 28. Total Annual Remediation Expenditures for Contaminated Sites (2005) in Europe as a % of GDP



Note: * Belgium: data refer only to Flanders.
Source: EEA CSI (2006).

Figure 29. % of Total Annual Remediation Expenditures for Contaminated Sites (2005) in Europe by Public and Private Sectors



Note: * Belgium: data refer only to Flanders. ** Italy: data refer only to region Piemonte.
Source: EEA CSI (2006).

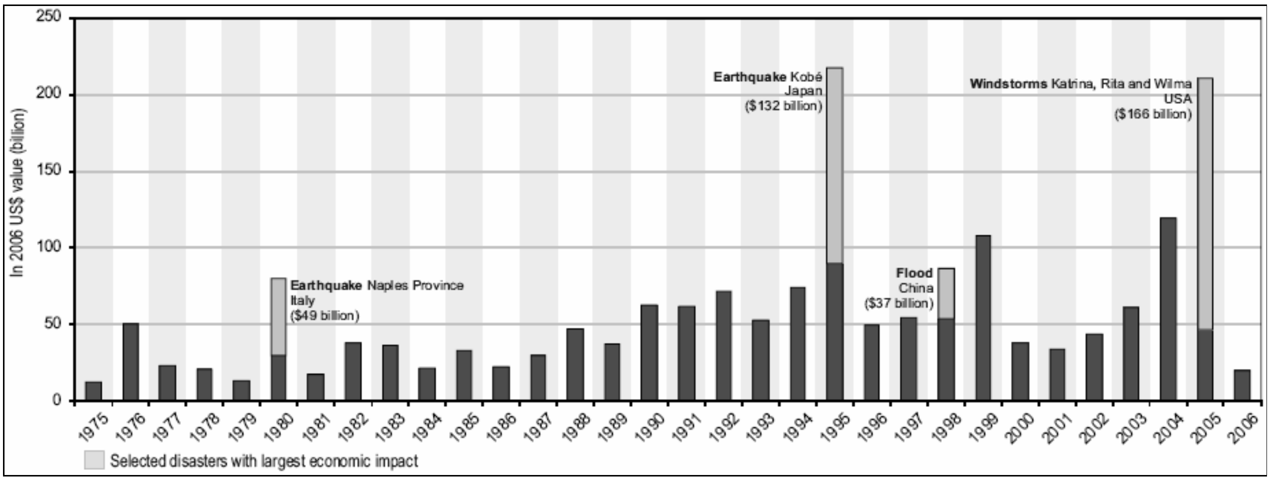
More significantly, the EU Environmental Liability Directive (2004/35/CE) imposes strict liability for "land contamination that creates a significant risk of human health being adversely affected as a result of direct of indirect introduction in, on, or under land of substances, preparations, organisms and micro-organisms". In the case of land contamination, the responsible party must remove the risk of human health being adversely affected, taking account of actual or planned future use (UK DEFRA, 2006). Conversely, for damages to water, the responsible party must return the environment to the condition it was before the event that gave rise to the damage. However, if the damage has been irreversible, or if it is not "cost effective" to return the resource to its baseline state, complementary remediation on another site must be undertaken, at the responsible party's expense. Given that it is not likely to be optimal to reduce the probability of irreversibility to zero, it is necessary to provide incentives for "compensatory" restoration.

Environment-related natural disasters

Available data indicate that there is an upward trend in the number of environment-related natural disasters and the magnitude of damages they inflict. Events such as windstorms, floods, and temperature extremes have typically been responsible for over 90% of the overall economic costs of extreme weather-related events each year over the period 1970-2005. In terms of insured losses, windstorms have contributed over 75% (and floods about 10%) of damages (OECD, 2006a).

World-wide, more than 255 million people were affected by (and 58,000 people died due to) natural disasters each year, on average, between 1994 and 2003. In 2003, 1 in 25 persons world-wide was affected by natural disasters. Damage caused by natural disasters was estimated at 67 billion USD per year, on average, during the decade 1994-2003 (Figure 30). The economic cost associated with natural disasters has increased 14-fold since the 1950s (Guha-Sapir *et al.*, 2004). Of course, only a proportion of these damages are related to environmental factors, and within this subset, only a proportion arise out of forces which are at least partly linked to human behaviour.

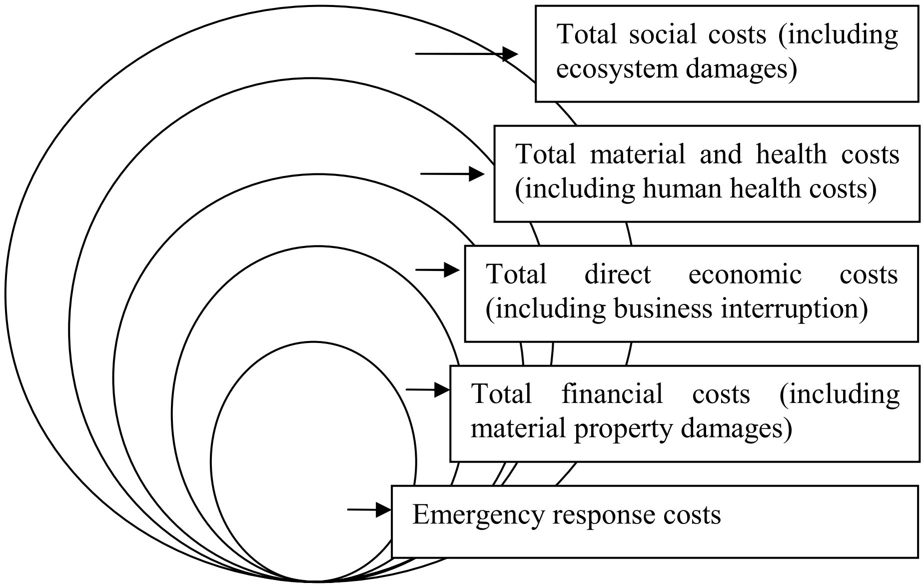
Figure 30. Annual Reported Economic Damages from Natural Disasters, 1975 – 2006 (in Billion USD, Using 2006 Prices)



Note*:* Epidemics and insect infestations not included.
Source: EMDAT (2007).

The costs associated with such events can be disaggregated into a number of different categories. The innermost circle in Figure 31 captures emergency response costs, which are relatively easy to determine. Moving outward, material property damages (capital equipment, housing, *etc*.) are added. Some of these may be insured; others not. These damages may result in significant disruption of economic activities, and this is captured in the next circle in the figure. The impacts on human health – such as physical injuries, disease associated with disruption in infrastructure, and mortality – are reflected in the next circle. And finally, in the outermost circle, ecosystem damages are included.

Figure 31. Social Costs of Inaction with Respect to Natural Disasters



The actual incidence of the costs outlined in Figure 31 depends very much on the damages covered by private insurance. This varies across countries and by type of disaster. The extent to which damages are insured will, of course, affect the incidence of the costs of inaction. For example, in many developing countries, the costs reflected in the first four circles will be borne almost entirely by the public sector (and development agencies and international donors).

In OECD countries, many of these costs will be recoverable from private insurance, which is often mandatory.[76] However, there is variation even within the OECD. In 2005, major catastrophes inflicted economic damages of $230 billion, $83 billion of which was covered by insurance ($45 billion of insured losses due to Hurricane Katrina alone) (Kunreuther and Michel-Kerjan, 2007).

Table 43 lists the twenty most costly events (in terms of insured losses) world-wide between 1970 and 2005. All of these events, except for the “9/11” attacks, were natural disasters. Among the top 19 natural disasters that occurred in the past 35 years, more than 80% were weather-related events, with nearly three quarters of the claims being concentrated in the US. The US has accounted for 4 times more insured losses than Europe ($320 billion in the US versus $80 billion in Europe) (OECD, 2006a).

76 In this case, the insurance coverage is purchased by the “victims”, whereas in the previous section (on industrial accidents), the insurance coverage is held by the “responsible” parties (*i.e.* maritime fleet owners or hazardous waste generators). The distinction arises out of the uncertainty with respect to the degree of causality in the case of natural disasters.

Table 43. The Twenty Most Costly Insured Events World-wide, 1970-2005 Insurance Losses

Rank	Event	Year	Insured loss (In USD m indexed to 2005)	Victims*	Area of primary damage
1	Hurricane Katrina	2005	45 000	1 326	US, Gulf of Mexico, Bahamas, North Atlantic
2	Hurricane Andrew	1992	22 274	43	US, Bahamas
3	Terror attack on WTC, Pentagon and other buildings	2001	20 716	2 982	US
4	Northridge Quake (M 6.6)	1994	18 450	61	US
5	Hurricane Ivan; damage to oil rigs	2004	11 684	124	US, Carribean *et al.*
6	Hurricane Rita; damage to oil rigs, floods	2005	10 000	34	US, Gulf of Mexico, Cuba
7	Hurricane Wilma; torrential rain, floods	2005	10 000	35	US, Mexico, Jamaica, Haiti *et al*.
8	Hurricane Charley	2004	8 272	24	US, Cuba, Jamaica *et al.*
9	Typhoon Mireille/No 19	1991	8 097	51	Japan
10	Winterstorm Daria	1990	6 864	95	France, UK, Belgium, NL *et al.*
11	Winterstorm Lothar	1999	6 802	110	France, Switzerland, UK *et al.*
12	Hurricane Hugo	1989	6 610	71	Puerto Rico, USA *et al.*
13	Hurricane Frances	2004	5 170	38	US, Bahamas
14	Storms and floods in Europe	1987	5 157	22	France, UK, NL *et al.*
15	Winter storm Vivian	1990	4 770	64	Europe
16	Typhoon Bart/No 18	1999	4 737	26	Japan
17	Hurricane Georges	1998	4 230	600	US, Carribean
18	Hurricane Jeanne ; floods and landslides	2004	4 136	3 034	US, Carribean ; Haiti *et al.*
19	Typhoon Songda/No 18	2004	3 707	45	Japan, South Korea
20	Tropical Storm Allison; heavy rain, floods	2001	3 475	41	US

Note: * Includes dead or missing.
Source: OECD (2006q), using data from Swiss Re and Insurance Information Institute.

The great disparity between North America and Europe in insured losses can be explained by the greater overall economic damages, as well as by higher insurance density (proportion of property which is insured and the maximum insurance coverage) in the US, compared with Europe. For example, the data suggest that the ratio of insured losses to overall losses has been about 38% in the US versus about 27% in Europe during the period 1980-2005 (Munich Re, 2006). However, these figures vary by incident. While insurance density in the US is thought to be in the region of 25–50% (Munich Re, 2006), in the case of Hurricane Andrew, the relevant figure was approximately 65%. For Katrina, it was 27–33% (Munich Re, 2006) (Table 44).

Table 45 lists the twenty worst catastrophes in terms of victims over the period 1970–2005. While a large percentage of these are by definition unrelated to factors which could be affected by environmental policy (*e.g.* earthquakes, volcano eruptions), the probability and severity of floods, cyclones and other extreme weather events may be affected by environmental policy factors. Interestingly, there appears to be no correlation between the most costly insured events (which occur primarily in developed countries) and the most deadly events (often in less-developed countries).

Table 44. Hurricane Katrina Insured Losses (Estimates in Billions US$)

	Low	High
Personal property lines	15.2	19.3
Residential Property	*14.0*	*17.0*
Personal auto	*1.0*	*2.0*
Personal watercraft	*0.2*	*0.3*
Commercial property lines	19.7	25.3
Commercial property (excl. off-shore)	*13.5*	*16.0*
Business interruption (excl. marine and energy)	*6.0*	*9.0*
Commercial auto	*0.2*	*0.3*
Marine and energy	4.0	6.0
Liability	1.0	3.0
Other	0.0	1.0
TOTAL	39.9	54.6

Source: Towers Perrin (2005).

The burden of the costs on the public sector is likely to be greater when there is limited insurance, as is often the case in developing countries. For example, the 1996 floods in China inflicted $24 billion in economic loss, but less than $0.5 billion (or 2.1%) was covered by insurance. The 1998 floods in China cost about $30 billion in economic loss, but only $1 billion (or 3.3%) were covered by insurance. Low insurance density has been observed also in industrialised countries where there are no minimum insurance requirements. The earthquake that devastated Kobe in 1995 cost $110 billion, but only $3 billion (2.7%) was covered by insurance (OECD, 2006a).

As noted above, the costs of inaction from uncertain events (such as natural disasters) are reflected in risk. This risk is a function of both hazard rate and vulnerability. Through policy action, governments can reduce both hazard rate and vulnerability to some extent. In the earlier discussion of industrial accidents, the focus was on the role that policy inaction can play in increasing hazard rates. In this Section, the focus is on factors affecting vulnerability, with three different examples: windstorms, floods, and extreme temperature events.[77] While the hazard rate for all three of these is affected by environment-related anthropogenic factors, in this Chapter, these are taken as given.[78]

77 The relative contribution of greenhouse gases and other anthropogenic factors to these events is a subject of continuing debate. According to a report published by the US National Academy of Science "greenhouse warming and other human alterations of the earth system may increase the possibility of large, abrupt, and unwelcome regional or global climatic events. ... Future abrupt changes cannot be predicted with confidence and climate surprises are to be expected" (http://books.nap.edu/openbook.php?record_id=10136&page=1). The recent IPCC report lists as 'more likely than not' the 'likelihood of a human contribution" to all three types of event. See IPCC WG1 (2007).

78 See also Chapter 3, this volume.

Table 45. The Twenty Worst Catastrophes in Terms of Victims (1970-2005)

Rank	Event	Year	Insured loss (In USD m indexed to 2005)	Victims*	Area of primary damage
1	Storm and flood catastrophe	1970	-	300 000	Bangladesh
2	Earthquake (M 7.5)	1976	-	255 000	China
3	Earthquake (Mw 9), tsunami in Indian Ocean	2004	2 068	220 000	Indonesia, Thailand *et al.*
4	Tropical cyclone Gorky	1991	3	138 000	Bangladesh
5	Earthquake (Mw 7.6); aftershocks, landslides, floods	2005	-	73 300	Pakistan, India *et al.*
6	Earthquake (M7.7); rock slides	1970	-	66 000	Peru
7	Earthquake (M 7.7)	1990	172	50 000	Iran
8	Earthquake (M 6.5) destroys 85% of Bam	2003	-	26 271	Iran
9	Earthquake (M7.7) in Tabas	1978	-	25 000	Iran
10	Earthquake (M6.9)	1988	-	25 000	Armenia, ex-USSR
11	Volcanic eruption on Nevado del Ruiz	1985	-	23 000	Colombia
12	Earthquake (M 7.5)	1976	257	22 084	Guatemala
13	Earthquake (ML 7.0) in Izmit	1999	1 173	19 188	Turkey
14	Dyke burst in Morvi	1979	-	15 000	India
15	Cyclone 05B devastates Orissa state	1999	117	15 000	India, Bangladesh
16	Flooding following monsoon rains in northern parts	1978	-	15 000	India, Bangladesh
17	Earthquake (Mw 7.7) in Gujarat	2001	110	15 000	India, Pakistan, Nepal *et al.*
18	Flooding in Bay of Bengal and Orissa state	1971	-	10 800	India
19	Floods, mudflows and landslides	1999	258	10 000	Venezuela, Colombia
20	Tropical cyclone in Bay of Bengal	1985	-	10 000	Bangladesh

Note: * Includes dead or missing.
Source: OECD (2006a), using data from Swiss Re and Insurance Information Institute.

Windstorms [79]

Economic damage caused by hurricanes in the US during the period between 1933 and 2005 reached, on average, $7.8 billion (in 2005 prices) or an equivalent of 0.062% of annual GDP. Individual events, however, can inflict damage of a much larger magnitude. For example, the economic impact of Hurricanes Andrew in 1992 and Katrina in 2005 amounted to 0.42% and 0.65% of annual GDP, respectively (Nordhaus, 2006).

Recent studies indicate that there has been an increase in tropical cyclone activity in the North Atlantic over the last three decades (Nordhaus 2006). Moreover, it has been estimated that hurricane "power", measured by the power dissipation index (PDI), has increased markedly since the mid-1970s (Emmanuel 2005). Nevertheless, it has been suggested that "despite the increase in overall hurricane activity, the US has not seen a significant resurgence of exceptionally strong [*i.e.* category 4-5] hurricane landfalls" (Blake *et al.* 2007:11) (Table 46). It therefore appears that the evidence is somewhat inconclusive.

[79] Windstorms may include a wide range of phenomena. This Section focuses on the impacts of *tropical cyclones*, also referred to as *hurricanes* or *typhoons.*

Table 46. Average Number of Tropical Cyclones which Reached "Storm", "Hurricane", and "Major Hurricane" Status

Period	Number of years	Avg number of tropical storms per decade	Avg number of hurricanes (cat. 1-5)	Avg number of major (cat. 3-5) hurricanes
1851-2006	156	8.7	5.3	1.8
1944-2006	63	10.6	6.1	2.7
1957-2006	50	10.7	6.0	2.4
1966-2006	41	11.1	6.2	2.3
1977-2006	30	11.4	6.3	2.5
1987-2006	20	12.6	6.8	2.9
1997-2006	10	14.5	7.8	3.6

Data source: Blake *et al.* (2007)

The number of hurricanes by itself is not indicative of the potential damages. For example, the occurrence of fewer hurricanes does not mean that there is a lesser threat of disaster. Historical records show that some of the most intense US hurricanes (1935) and some of the costliest ones (*e.g.* Hurricane Andrew in 1992), occurred in years which had much below-average hurricane activity (Blake *et al.*, 2007).

So what, then, determines the magnitude of the costs of a disaster? As discussed earlier, much of the risk associated with a natural hazard is related to factors other than frequency and intensity. According to Nordhaus (2006), the vulnerability of a region to hurricanes depends on factors such as the location of economic activity, the total output (GDP), the capital intensity of output, and the geographical features of the affected areas.

The lower death tolls recorded in recent years (with the exception of 2005) are partly the result of relatively few major hurricanes striking the most vulnerable areas (Blake *et al.*, 2007). At the same time, a hurricane of a medium intensity, such as Katrina, became the most costly hurricane in recent history. However, Katrina was so costly not so much because of its intensity, but because it hit an economically vulnerable region in the US (Nordhaus, 2006)[80].

Table 47 lists the most costly mainland US tropical storms between 1900 and 2006. Three lessons can be drawn from the data:

- There is a weak relationship (correlation = 0.23) between the category of the storm and the damage inflicted, suggesting that hurricane damage depends primarily on the vulnerability of the location of landfall.
- Most (26 of the top 30) costly hurricanes occurred in the period after year 1960, suggesting that one of the reasons for the observed rising hurricane damages may be intrinsic to general growth in prosperity, as well as to growing coastal development.
- There is a weak relationship (correlation = 0.16) between property damage and the death toll inflicted by hurricanes, suggesting that the most costly hurricanes are not necessarily the deadliest ones.[81]

80. Katrina killed 1,300 people and forced 1.5 million people to evacuate the affected area.

81 This calculation is based on the available data, excluding Hurricane Katrina, which is frequently qualified as an outlier (see for example Nordhaus, 2006).

Table 47. The Thirty Costliest Mainland US Tropical Cyclones, 1900-2006 Property Damages

Rank	Hurricane	Year	Category	Damage (million 2006 US$)*	Deaths
1	KATRINA	2005	3	84,645	1500
2	ANDREW	1992	5	48,058	-
3	WILMA	2005	3	21,527	-
4	CHARLEY	2004	4	16,322	-
5	IVAN	2004	3	15,451	25
6	HUGO	1989	4	13,480	-
7	AGNES	1972	1	12,424	122
8	BETSY	1965	3	11,883	75
9	RITA	2005	3	11,808	-
10	CAMILLE	1969	5	9,781	256
11	FRANCES	2004	2	9,684	-
12	DIANE	1955	1	7,700	184
13	JEANNE	2004	3	7,508	-
14	FREDERIC	1979	3	6,922	-
15	New England	1938	3	6,571	256
16	ALLISON	2001	TS	6,414	41
17	FLOYD	1999	2	6,342	56
18	Northeast US	1944	3	5,927	64
19	FRAN	1996	3	4,979	26
20	ALICIA	1983	3	4,825	-
21	OPAL	1995	3	4,758	-
22	CAROL	1954	3	4,345	60
23	ISABEL	2003	2	3,985	-
24	JUAN	1985	1	3,417	-
25	DONNA	1960	4	3,345	50
26	CELIA	1970	3	3,038	-
27	BOB	1991	2	2,853	-
28	ELENA	1985	3	2,848	-
29	CARLA	1961	4	2,604	46
30	DENNIS	2005	3	2,330	-

Notes: * Includes property damage only. Adjusted for inflation using the 2006 deflator for construction.
TS = only of tropical storm intensity.
- Indicates no data or less than 25 deaths.

Data Source: Blake *et al.* (2007).

This highlights the importance of coastal growth as a key determinant of vulnerability. It is likely that if more coastal areas are developed, hurricane damages will keep rising, simply because more people and more property will be present in the affected areas. Moreover, it is likely that even weaker hurricanes and tropical storms may, in the future, inflict major catastrophes on the affected communities.

Therefore, greater effort in developing preparedness plans (warning and evacuation plans), changes in building practices (regulations of coastal development), and addressing factors related to development of insurance markets are needed. Specifically, in the face of the growing risks of natural disasters, governments have the following policy options available:

- Place more restrictions on coastal development, in order to reduce potential future property damage and human losses; and,

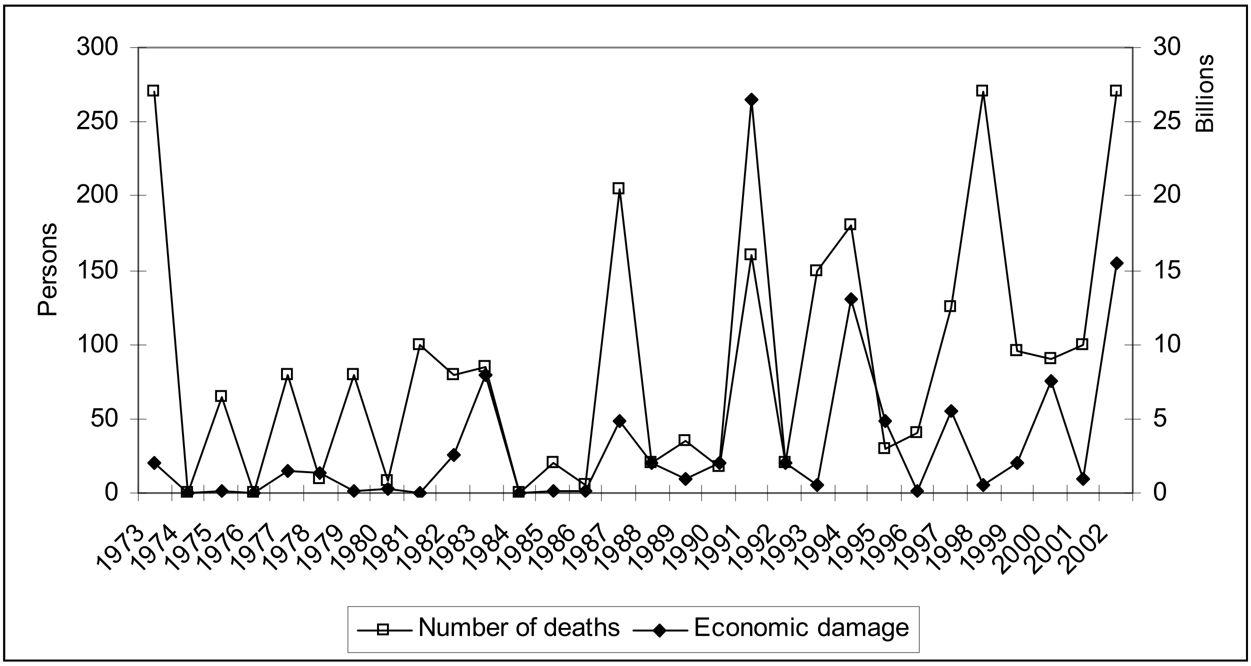
- Institute minimum insurance requirements and assist in developing insurance markets, in order to increase insurance coverage of coastal properties.

In the absence of such measures, governments will face pressure to provide public funding for rescue operations and post-disaster reconstruction efforts.

Floods

While hurricanes can result in extensive flooding – due to storm surge which temporarily raises sea level – flooding of river embankments may be equally disastrous. Figure 32 provides time series data linking death tolls and economic damages from flooding in Europe. However, preventive measures can reduce the impact of such events. The World Bank (2004b) estimated that $3.15 billion spent on flood control in China between 1960 to 2000 averted losses of about $12 billion.

Figure 32. Death Toll and Economic Damages Caused by Flooding in Europe, 1973-2002 (Billions of 2002 EUR)



Source: Data from Hoyois and Guha-Sapir (2003).

Other evidence suggests that mitigation is often a cost-effective way of reducing risks from natural hazards. For example, in an analysis of a statistically representative sample of FEMA (US Federal Emergency Management Agency) mitigation grants awarded between 1993 and 2003 showed that 1 dollar of mitigation expenditure potentially saves it an average of 3.65 dollars, reflected in avoided post-disaster relief costs and increased federal tax revenues (US NIBS, 2005). Overall, the study concluded that mitigation is sufficiently cost-effective to warrant federal funding; and that it is most effective when it is carried out on a comprehensive, community-wide, long-term basis, with a focus on building resilient communities[82] (Table 48).

82 The costs considered in the analysis included both the federal and local shares of costs. The benefits were defined broadly, including reductions in property damage, direct and indirect business interruption losses, non-market damage, human losses, and reduced costs of emergency response.

Table 48. Benefits and Costs of Mitigation by Hazard

Hazard	Cost ($M)	Benefit ($M)	Benefit-Cost Ratio
Earthquake	947	1,392	1.5
Wind	374	1,468	3.9
Flood	2,217	11,189	5.0
Total	3,538	14,049	4.0

Source: US NIBS (2005).

The variation in benefit-cost ratios across hazards in Table 48 is mostly due to the different types of avoided damage, which can be characterised by different degrees of variability and uncertainty over hazard frequencies. For example, in the sample of grants analysed in the study, 95% of flood benefits were attributable to avoided property damage (with only 3% for casualty reduction in contrast to 60% for casualty reduction for wind hazards). Since factors affecting structures have, in general, lower variability and there is less uncertainty associated with flood frequency distributions than those of other hazards (history of vulnerabilities in floodplains, recurrence of floods in a given location, *etc*.), it is easier to protect structures than to reduce casualties. As a result, cost effectiveness of measures to reduce property damage is higher than for reducing casualties (US NIBS, 2005).

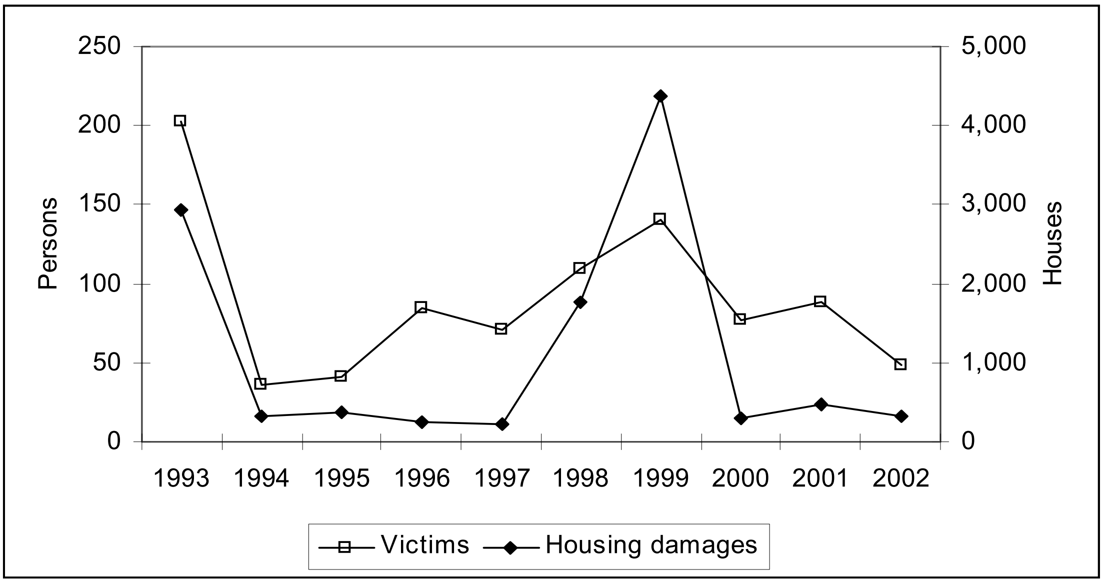
Flooding is also a serious natural hazard in Japan (Figure 33), where approximately 49% of the land where the population lives and 75% of total property value is located in flood plains (Zhai *et al.*, 2003). Over the past several decades, Japan has adopted various risk mitigation measures (*e.g.* dam and dyke construction) to reduce the risks from flooding. However, although the mitigation measures have resulted in a dramatic decrease in human losses from flood hazards, economic losses have not decreased to the same extent (Zhai *et al.*, 2003).

According to flood disaster statistics, flood losses in Japan increased markedly before the 1950s, and then began to decrease in 1960s. Since the 1970s, there has been no decrease, despite continuous increases in flood prevention investments (Figure 34).

This pattern can be partly explained by increasing population (from 89 million in 1955 to 127 million in 1999) and increases in wealth (fixed assets increased from 2 trillion yen in 1955 to 139 trillion yen in 1996, using 1980 prices) (Zhai *et al.* 2003). However, the effectiveness of further flood prevention measures on the scale previously undertaken has been questioned (Zhai *et al.* 2003).

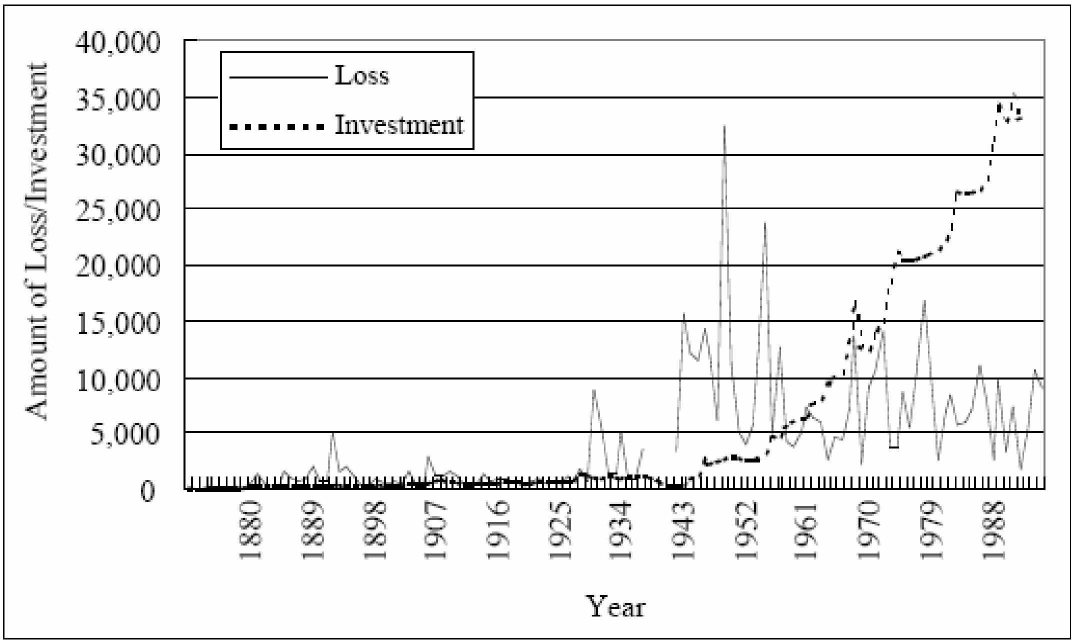
Since the 1980s, the returns on investment in flood prevention have been less impressive. Figure 35 shows the ratio of total benefits to total investment costs. The ratio drops sharply from more than 200 during the early 1960s, through 2-10 during 1970s, through 1-2 during 1980s, to less than 1 after 1988. If human and intangible losses (*e.g.* pain and suffering) are taken into account as well, there is a slight vertical shift upward. However, the shape of the curve remains fundamentally the same (Figures 35 and 36). Again, the ratio drops below unity after 1988.

Figure 33. Damages by Storms and Floods in Japan, 1993-2002



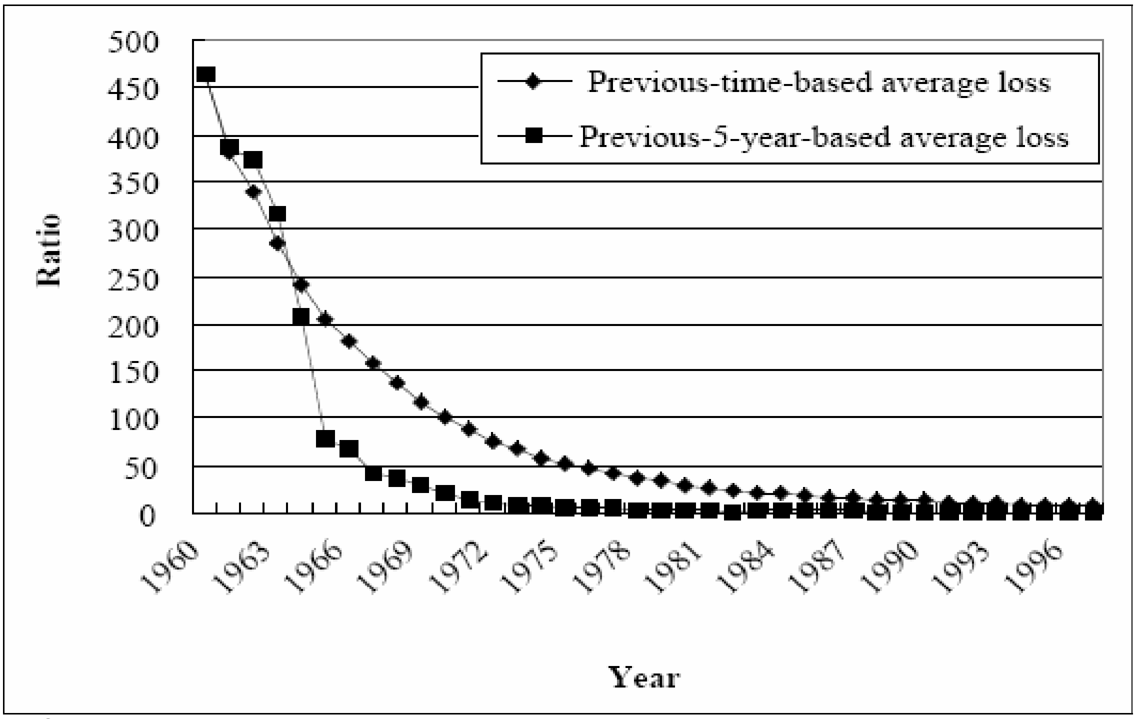
Notes: Victims = number of fatalities and missing persons.
Housing damages = number of houses completely or partially collapsed.
Source: Data from Japan Fire and Disaster Management Agency (www.fdma.go.jp).

Figure 34. Flood Damage and Flood Prevention Investment in Japan, 1872-1995



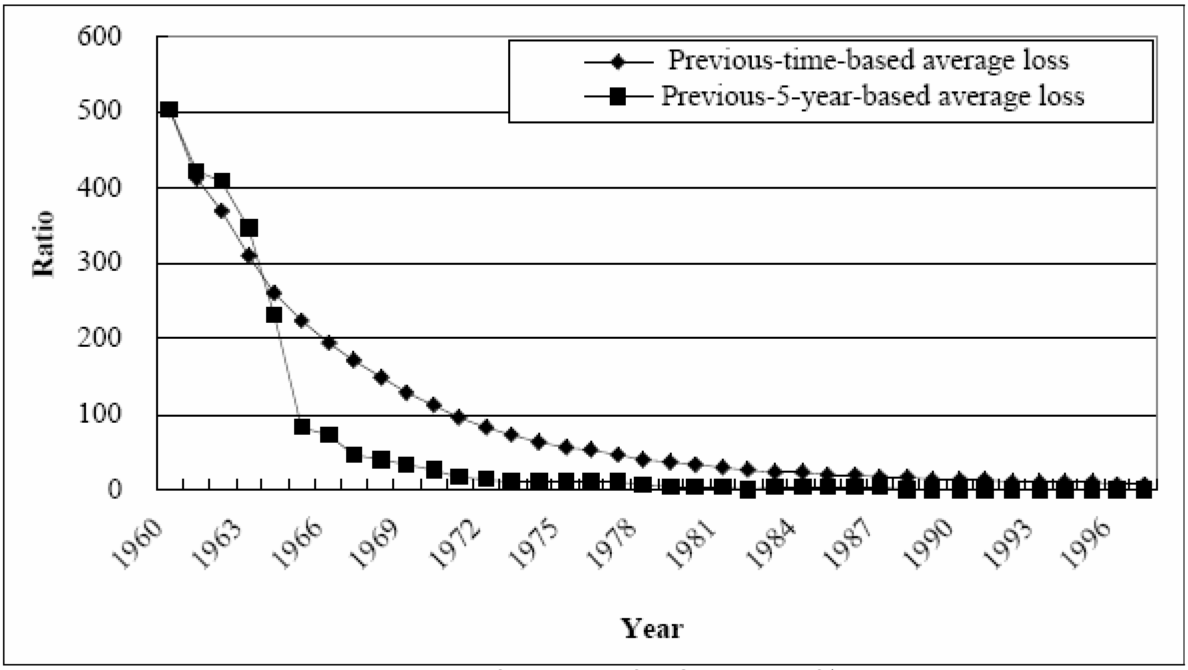
Note: Unit: 100 million yen at 1995 prices.
Source: Zhai *et al.* (2003), using data from Ministry of Land, Infrastructure and Transport.

Figure 35. Ratio of Total Benefit to Total Cost (Material Losses)



Source: Zhai *et al.* (2003).

Figure 36. Ratio of Total Benefit to Total Cost (Material, Human and Intangible Losses)



Note*:* Human loss evaluated using a value-of-a-statistical-life (VSL) estimate of $9.2 million.
Source: Zhai *et al.* (2003).

In sum, while initial flood prevention measures may have greatly contributed to the reduction of flood hazards, every additional flood protection measure will be less efficient at reducing these risks. Thus, it is necessary to compare the costs of additional investment with the benefits in terms of its contribution to flood prevention. Moreover, excessive levels of flood protection may actually lead to perverse outcomes. Under certain conditions, too much flood protection may actually increase the risk of flooding (reduced water retention capacity, wetland destruction, constrained flood plain, *etc.*). Addressing costs of inaction with respect to the "hazard" may be more efficient.

Heat waves

In the summer of 2003, Europe experienced an exceptionally long and severe heat wave[83] with enormous adverse social, economic, and environmental effects, including death of thousands of vulnerable elderly people, economic damages due to power cuts and transport restrictions, lost agricultural production, destroyed forests due to wildfires, and effects on water ecosystems and glaciers (UNEP 2004). According to INSERM (the French National Institute of Health and Medical Research), excess mortality during summer 2003 exceeded 70,000 deaths in Europe, with the highest death toll being recorded in France (19,490 excess deaths) and Italy (20,089 excess deaths) (INSERM, 2007).

Table 49 gives estimates of excess mortality from the heat wave. The consequences were probably underestimated in many countries, at least based on the first estimates (EuroSurveillance, 2005). The most important risk factors affecting the vulnerability of people to heat have been identified to include age (particularly those over 75 or 80 years), age-associated factors (loss of autonomy, social isolation), and location (living directly below the roof or in a "heat island") (EuroSurveillance, 2005).

Table 49. Heat-associated Excess Mortality in Summer 2003 in Selected Countries (Number of Excess Deaths)

Country	Estimate	
France	14,800 [a]	+ 60%
Italy	19,780 [b,c]	
England and Wales	2,139 [d]	+ 16%
Spain	6,595 – 8,648 [e]	+ 8-11%
Netherlands	1,400 – 2,200 [f]	+ 3-5%
Germany	1,410 [g]	
Portugal	1,316 [h]	+ 38%
Belgium	1,250	
Switzerland	975	
Total of above countries	49,665 - 52,518	

Note: Data refer to a 3-month period June-August 2003, unless stated otherwise.
[a] Data are for 1-20 August 2003.
[b] A country-wide estimate of the Italian National Institute of Statistics for June-September 2003.
[c] The estimate for the cities of Bologna, Milan, Rome, and Turin for August 2003 alone is 2,255 (+23%) excess deaths.
[d] Data are for 4-13 August 2003.
[e] The estimate for August 2003 alone is 3,574 - 4,687 (+17%) excess deaths.
[f] The estimate for the period 31 July - 13 August 2003 is 500 excess deaths.
[g] Data are for 1-24 August 2003 and for the Region of Baden-Württemberg only.
[h] Data are for 15-28 July 2003.
Source: Adapted from EuroSurveillance (2005).

In the US, heat waves kill an estimated 1500 people every year (IFRC, 2004). For comparison, the combined death toll from hurricanes, tornadoes, earthquakes and floods is less than 200. The heat wave that hit Chicago in 1995 killed 739 people. In Oceania, the effects of heat waves are also significant, but

[83] According to Juerg Luterbacher (University of Bern, Switzerland), the summer of 2003 was the hottest in Europe for at least 500 years (IFRC, 2004).

not widely-recognised. According to a report (McMichael *et al.*, 2003) prepared for the Australian Department of Health and Ageing extreme temperatures currently contribute to the deaths of some 1100 people aged over 65 each year in 10 Australian and 2 New Zealand cities. This is projected to rise to between 4,300 and 6,300 by 2050, but much of the increase is attributable to an ageing population and not temperature increases *per se*.

Kysely (2004) analysed mortality data for the period 1982-2000 in the Czech Republic and found that the mean relative rise in total mortality during heat waves was 13% (with increases in mortality of up to 37% for individual events). The study found that the "mortality displacement effect" was important, because mortality tended to be lower than expected after hot periods. When the displacement effect[84] is taken into account, the mean net mortality change due to heat waves was estimated to be about 1%.

Laschewski and Jendritzky (2002) found increases in mortality of up to 25% in the German region of Baden-Württemberg during heat waves, but the net mortality change due to heat waves was about 0.2%. Huynen *et al.* (2001) found the mean relative increase in the total mortality of 12% during heat waves in the Netherlands (with the largest excess of about 24% for an individual event), but with inconclusive results about the short-term mortality displacement.

Overall, the impacts of heat stress on mortality are most pronounced for the elderly, children, and people whose health has already been compromised, particularly due to cardiovascular, cerebrovascular, and respiratory diseases (Kysely 2004).[85] Various studies have also reported that females are more sensitive to heat stress than males (*e.g.*, Díaz *et al.*, 2002a,b; Kysely, 2004; Mackenbach *et al.*, 1997; Rooney *et al.*, 1998).

To evaluate the costs of inaction associated with adaptation policies that reduce heat-related excess mortality (and of other environmental policies, such as those that limit exposure to air pollutants), estimates of value of a statistical life (VSL) obtained from contingent valuation studies (using willingness to pay estimates) or labour market studies (using hedonic prices) must be used. In one Italian study, the value of a statistical life (VSL) ranged from EUR 0.257 million to over EUR 5.8 million, based on willingness-to-pay estimates for reducing risks of dying for cardiovascular and respiratory causes, the most important causes of premature mortality associated with heat wave and air pollution (Alberini and Chiabai, 2006).

On top of the catastrophic death toll, Europe suffered also significant financial losses due to the 2003 heat wave. The total losses are estimated to have exceeded 13 billion Euros. This includes the losses of the arable and livestock sector due to droughts and fires (Table 50). In the forestry sector, more than 25,000 fires were recorded destroying nearly 650,000 hectares, most of it in Portugal (390,146 hectares; 5.6% of its forest area; with financial impact estimated at 1 billion EUR) (UNEP 2004).

84 The "displacement effect" refers to situation when some of the heat-related deaths are short-term displacements of the deaths of critically ill people who would have died soon thereafter, even in the absence of oppressive weather conditions (see *e.g.* Kysely, 2004; Hyunen *et al.*, 2001; Braga *et al.*, 2002).

85 Historically, cardiovascular diseases have accounted for 13-90% of the increase in mortality during and following a heat wave; cerebrovascular disease has accounted for 6-52%; and respiratory disease has accounted for 0-14% (Kilbourne, 1997).

Table 50. Financial Impact of the Summer 2003 Drought and Fires on the Agricultural and Forest Sectors

	Financial impact (millions EUR)	
Austria	197	
- fodder		*150*
- cereals		*30*
France	4000	
- beef		*1500*
- maize		*265*
- fruits		*515*
- livestock		*100*
Germany	1500	
- fodder		*650*
- cereals		*389*
- potatoes		*275*
- sugar		*100*
Italy	4-5000	
Portugal	1030	
- forest fires		*1030*
Spain	810	
- arable crops		*710*
- livestock		*100*
Estonia	7	
Hungary	453	
Slovakia	143	
CEEC	603	
EU15 and candidate countries	13000	

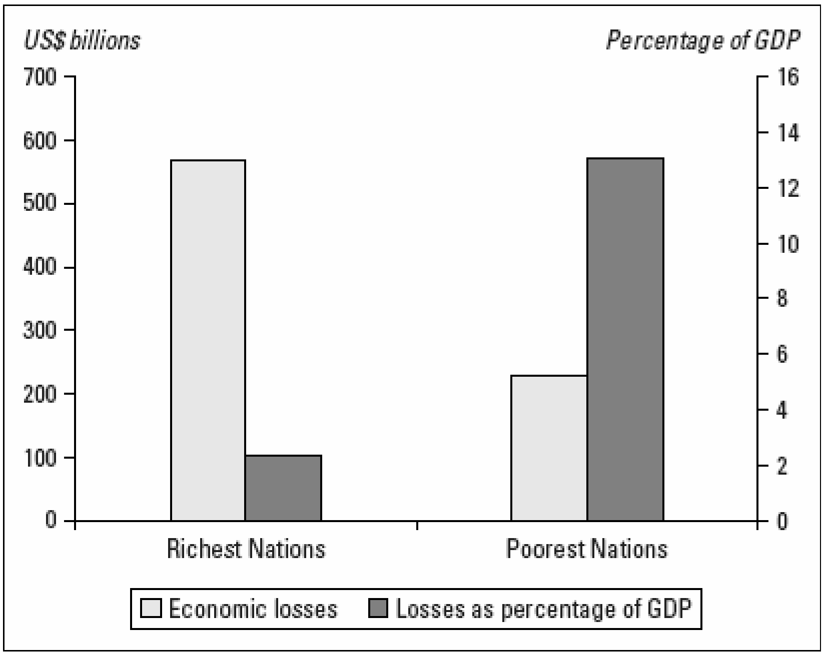
Data source: COPA-COGECA (2003).

The heat wave also caused losses due to power cuts and transport restrictions. For example, many of France's nuclear reactors had to shut down due to low river water levels, which were insufficient for cooling, or due to temperatures of the returning cooling water in excess of environmental safety levels. In addition, demand for electricity soared due to increased consumption for air-conditioning and refrigeration. As a result, France (Europe's major electricity exporter) cut its power exports by more than a half (UNEP, 2004).

Natural disasters and economic development

There is a close link between poverty and disasters. According to one report (IFRC, 2001), 98% of the 211 million people affected by natural disasters each year from 1991 to 2000 were from developing nations. Relative economic losses due to natural disasters disproportionately affect the poor and the undeveloped. According to the World Bank (2006c), more than 90% of natural disaster-related deaths occur in developing countries. Even though the absolute magnitude of economic losses is far greater in developed countries, the size of losses relative to the volume of total output in developing countries far exceeds those in developed countries (Figure 37).

Figure 37. Disaster Losses in the Richest and Poorest Countries, 1985-1999



Source: UN/ISDR (2004).

Disasters are thus a major threat to economic development. Damage due to natural disasters may constitute between 2% to 15% of an exposed country's annual GDP (World Bank, 2004b). For example, between 1990 and 2000, the damage from natural disasters amounted to, on average, 1.8 and 2.5% of GDP in Argentina and China, and as much as 12.58 and 15.6% in Jamaica and Nicaragua, respectively (Table 51).

Table 51. Damages Due to Natural Disasters as % of Country's Annual GDP, 1990-2000

Argentina	1.81%
Bangladesh	5.21%
China	2.50%
Jamaica	12.58%
Nicaragua	15.60%
Zimbabwe	9.21%

Source: World Bank (2004b).

Losses for individual events can be even more telling. For example, in Honduras, Hurricane Mitch caused losses equal to 41% of GDP, or an equivalent of 292% of the government's annual tax revenue (World Bank, 2004b). The enormous difference between the impacts of natural disasters in developed, versus developing countries, is also evident when it comes to human losses (Table 52).

Table 52. Comparing the Human Impact of Natural Disasters Between the 10 Richest and the 10 Poorest Countries

Country	GDP (US$) per capita 2002	Annual average victims per 100,000 population (1974-2003)	Country	GDP (US$) per capita 2002	Annual average victims per 100,000 population (1974-2003)
Luxembourg	44,000	0	Somalia	550	2,701
US	37,600	59	Sierra Leone	580	155
Norway	31,800	5	Burundi	600	674
Switzerland	31,700	2	Congo, RD	610	114
Ireland	30,500	4	Tanzania	630	1,531
Canada	29,400	72	Malawi	670	8,748
Belgium	29,000	2	Afghanistan	700	1,120
Denmark	29,000	0	Eritrea	740	6,402
Japan	28,000	182	Ethiopia	750	5,259
Austria	27,700	29	Madagascar	760	2,090

Source: Guha-Sapir *et al.* (2004).

Summary

The costs of inaction with respect to environment-related industrial accidents and natural disasters are an issue of increasing importance, with economic impacts for OECD and non-OECD countries. Such events are uncertain in many senses:

- the timing, frequency and severity of these events are uncertain;
- the causal factors which are responsible for their occurrence are uncertain; and
- the damages associated with them are uncertain.

While it is not economically efficient (or even feasible in most cases) to reduce the risk of these "events" to zero, there are two ways in which governments can reduce their risk. First, they can introduce policies which encourage investment in measures which reduce the *hazard rate* (*i.e.* probability and severity of an event occurring). Second, they can introduce policies which encourage investment in measures which reduce *vulnerability* (*i.e.* the costs associated with such an event should it arise).

Inaction with respect to these two sets of measures results in a variety of different costs, including: emergency response costs, remediation costs, material damages, human health losses, and ecosystem damages. While the first of these are generally easier to assess, they represent a significant underestimate of the costs of inaction with respect to preventive activity.

The evidence presented indicates that these costs can be considerable. While "environmental" factors related to human behaviour (such as greenhouse gas emissions) are not the only contributors to such costs, they can be an important contributing factor. Moreover, in many cases, the *ex ante* costs of prevention and preparedness can be much less than the costs of *ex post* remediation and restoration. This is particularly true when some of the damages which arise are irreversible, or only reversible at very significant cost.

CHAPTER 5. COSTS OF INACTION WITH RESPECT TO NATURAL RESOURCE MANAGEMENT

Introduction

Natural resources can be distinguished between those which are renewable (forestry, fisheries, *etc*.) and those which are non-renewable (oil, coal, *etc*.). A distinction can also be drawn between living (renewable) and non-living (non-renewable) resources, with the latter also sometimes being characterised as "exhaustible" resources. (Freshwater is generally classified as a renewable resource.) In this Chapter, the cases of marine fisheries and groundwater are examined. While the former is unquestionably a "renewable" resource, the latter has an ambiguous character. Although some groundwater is replenished very quickly, in other cases, recharge can only take place over millennia, with the implication that groundwater is best understood as being analogous to mineral deposits and fossil fuels.

In both of the cases examined here, inefficient management arises out of the practical difficulties associated with excluding access to potential users. In such a context, the rate of exploitation will be excessive, and socially inefficient. In general terms, policy "inaction" in this context will involve situations in which access to the resource is not sufficiently controlled. Constraining access can be done directly (*i.e.* through property rights creation), or indirectly (*i.e.* regulatory or financial measures).

The costs of unsustainable use of natural resources can be considerable. Most obviously, this will include the direct costs associated with the loss of the resource in question. For instance, exploiting a fish stock to economic extinction will result in the loss of commercial yields forever. This can also have important indirect impacts on local communities and the wider economy. Given the importance of some natural resources (*e.g.* water) to economic development, significant public expenditures will be incurred to mitigate the welfare impacts of unsustainable resource exploitation. And finally, there are likely to be a wide variety of costs associated with impacts on non-use values, such as impacts on ecosystems which are not reflected in terms of impacts such as lost resource productivity.

Marine capture fisheries

Introduction

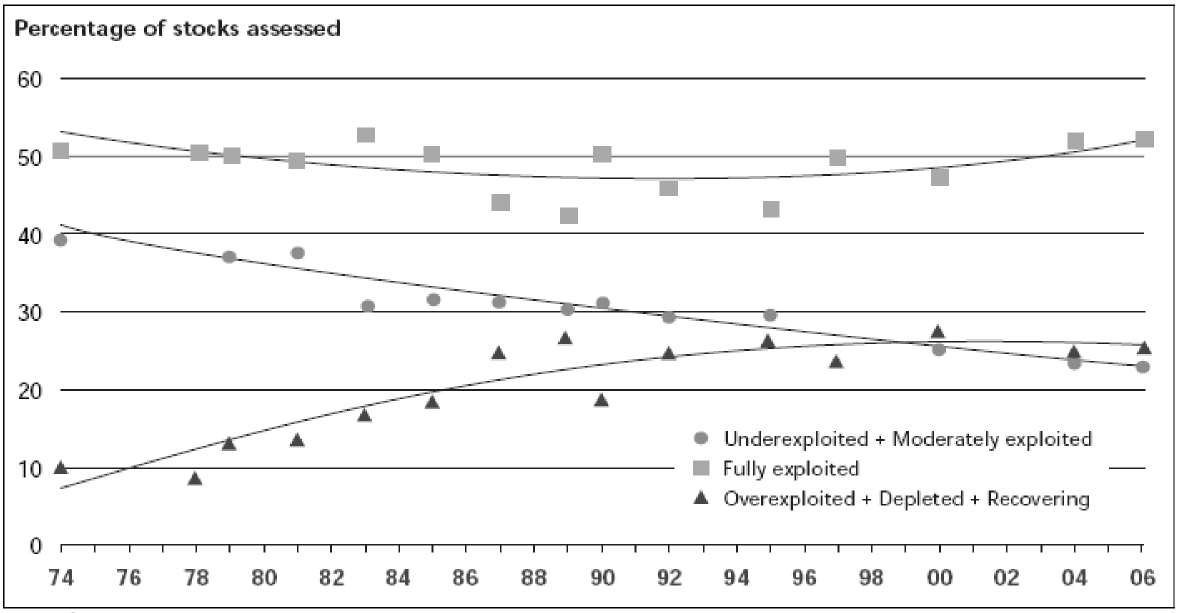
The fisheries sector is an important source of employment -- about 40 million fishers and fish farmers depend on fisheries worldwide (FAO, 2005). An overwhelming majority of these people (about 95%) are in developing countries (FAO, 1999). In many countries, fish is an essential part of the diet. For example, fish provide 22% and 19% of animal proteins consumed in Asia and Africa, respectively (FAO, 2005). The recreational opportunities associated with fishery resources also contribute to the livelihoods of coastal or island communities. The impacts of fisheries on aquatic ecosystems are being increasingly recognised. For these and other reasons, it is important that fishery resources be managed sustainably. Unsustainable fisheries management can have significant economic consequences. This Section discusses the costs of policy inaction with respect to fisheries management, with a primary focus on *marine capture fisheries*.

Status of world fisheries

According to the FAO (2007), the exploitation of the world marine fishery resources intensified rapidly during the 1970s and 1980s. Figure 38 illustrates that, although the proportion of fully-exploited stocks has remained more-or-less constant over the last three decades (at about 50%), there has been a notable increase in the proportion of over-exploited and depleted stocks (from 10% in 1974 to 25% in 2005).

According to FAO, "the maximum wild capture fisheries potential from world's oceans has probably been reached."

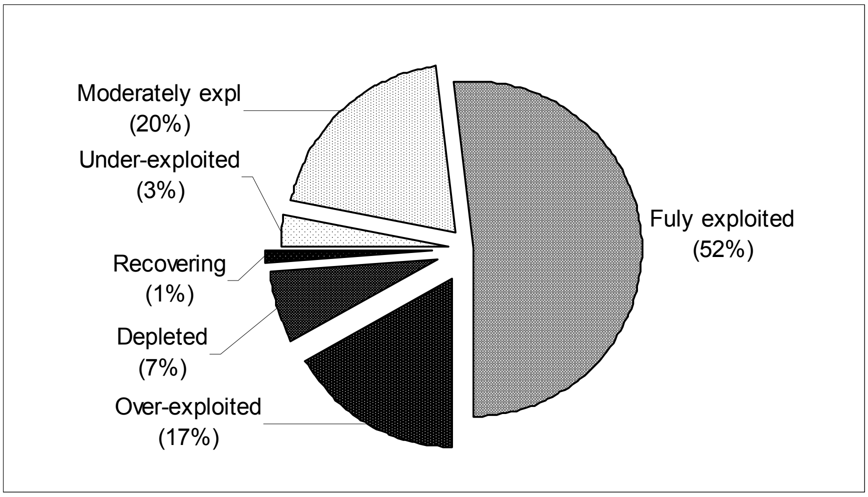
Figure 38. Global Trends in the State of World Marine Stocks (1974-2006)



Source: FAO (2007).

In 2005, less than one-quarter of the stocks being monitored by FAO were under-exploited (3%) or moderately-exploited (20%). Half of the stocks (52%) were being fully-exploited, therefore producing catches that were at or close to their maximum sustainable limits, leaving no room for further expansion. Most importantly, one-quarter of the stocks were being either over-exploited (17%), depleted (7%), or recovering from depletion (1%) (Figure 39).

Figure 39. Status of World Fish Stocks (2005)



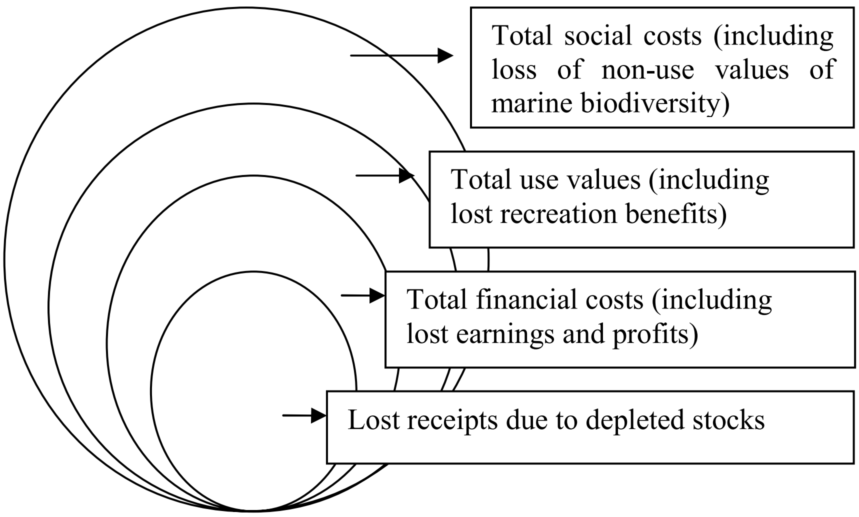
Source: Data from FAO (2007).

Policy inaction in the context of fisheries management

"Inaction" in the context of fisheries management can be best described as unsustainable resource management (*i.e.* where the stock is being exploited at a rate which is greater than that which can be supported). In practice, almost all fisheries are subject to some kind of regulation. Regulation of fisheries typically involves some constraints on (i) fishing gear restrictions; (ii) spatial and/or temporal restrictions on fishing; and (iii) volume restrictions on fish harvest and fishing effort. If the combination of regulatory measures in place is not sufficient to ensure sustainable resource management, the economic consequences can be considerable.

Figure 40 illustrates the kinds of costs arising from unsustainable fisheries management. The inner circle represents essentially the direct economic consequences of inaction – lost receipts of fishers and vessel owners from falling catches, caused by stock depletion. The next circle represents the direct as well as the more indirect economic consequences – lost earnings of workers and foregone profits of fish-processing and related industries. The next circle outward includes all use values – the aforementioned costs as well as costs which can be difficult to value due to their non-market characteristics, such as reduced recreational opportunities. And finally, the outer circle represents all impacts, including the costs associated with damages to marine ecosystems which are reflected in terms of non-use values.[86]

Figure 40. Costs of Inaction with Respect to Fisheries Management



Evidence of inaction (unsustainability)

While precise information on the status of fish stocks is complicated by the nature of the resource, there is some evidence of unsustainable fisheries management. A review of selected examples of individual fish stocks is illustrative.

86 The costs associated with damages to marine ecosystems and biodiversity which relate to use values are represented in the next circle inward.

Status of the world's major fish stocks

According to FAO (2007), most of the stocks of top ten species[87] (which account for about 30% of the world capture volume) are fully exploited or over-exploited, and therefore cannot be expected to produce further increases in catches. While the top ten species (by volume harvested) are not necessarily the most valuable ones, information on the assessment of fish stocks at shadow prices (or even at ex-vessel market prices) is not readily available. In the mid–1990s, it was estimated that catches of 77% of the marine species monitored by FAO (including 660 species by fishing area) had reached or exceeded sustainable levels (Table 53) (FAO, 1996).

Table 53. Global List of Fish Stocks Ranked as "Depleted"

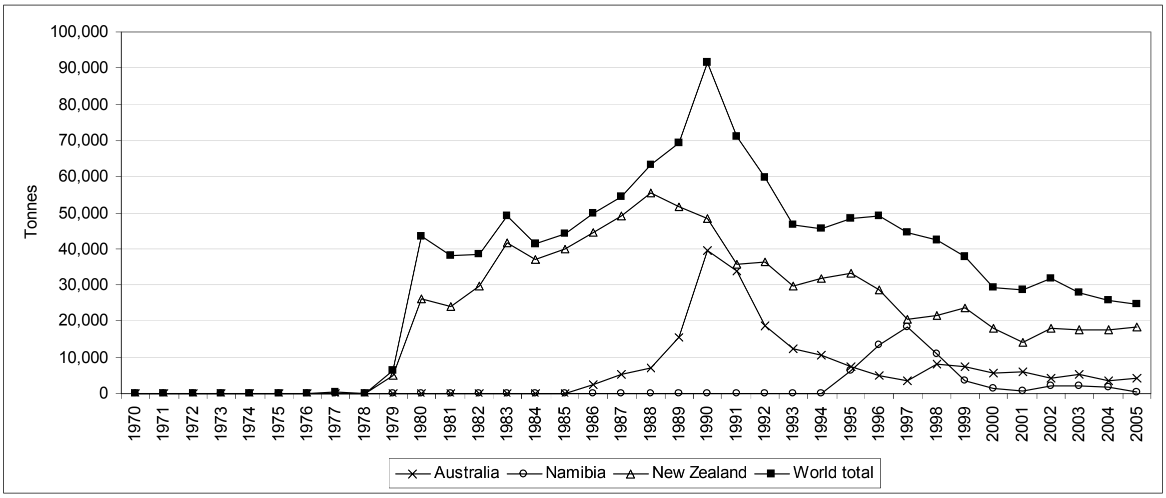
Stock	Status	Stock	Status
Northwest Atlantic (FAO area 21)		*Mediterranean/Black Sea (37)*	
Atlantic cod	D	Albacore	F-D
Haddock	D	Atlantic Bluefin Tuna	D
		Atlantic Bonito	F-D
Northeast Atlantic (FAO area 27)		Azov Sea Sprat	D
Atlantic cod	O-D	European Sprat	D
Atlantic salmon	F-D	Sardinellas	U-D
Haddock	O-D	Pontic Shad and other Shads	D
Other cods, hakes, and haddocks	F-D	Whiting	F-D
Salmons, trouts, smelts	F-D		
Whiting	F-D	*Northeast Pacific (67)*	
		North Pacific Hake	U-D
Southwest Atlantic (41)		Shrimps, prawns, *etc*.	F-D
Argentine hake	O-D		
		Southeast Pacific (87)	
Western Atlantic (31,41)		Eastern Pacific Bontito	O-D
Bluefin Tuna	D	South Pacific Hake	F-D
Indian Ocean (51,57,58)		*Southern Ocean (48,58,88)*	
Southern Bluefin Tuna	D	Antarctic Rockcods	D
		Blackfin Icefish	D
Pacific Ocean (61,67,71,77,81,87)		Patagonian Toothfish	F-D
Southern Bluefin Tuna	D	Mackerel Icefish	D
Southeast Atlantic (47)			
Geelbek Croaker	D		
Red Steenbras	D		

Notes: D: depleted; O-D: ranging from over-exploited to depleted; F-D: ranging from fully exploited to depleted; U-D: ranging under-exploited to depleted. Stock assessments based on 2004 data.
Source: FAO (1996).

Historically, there have been several instances of stock over-exploitation. The threat of over-fishing is particularly important in the case of deep-sea (deep demersal) fisheries. Deep-sea fishes are generally very long-lived (more than 100 years, in some cases), late to mature, slow growing, of low fecundity, and prone to formation of dense aggregations for spawning and/or feeding (Lack *et al*., 2003). As a result, they are relatively unproductive, highly vulnerable to over-fishing, and potentially slow to recover from the effects of over-exploitation. For example, many of the stocks of *Orange Roughy* have been depleted in the course of two decades (Figure 41).

87 The top ten species here include anchoveta, Alaska pollock, blue whiting, Atlantic herring, Japanese anchovy, Chilean jack mackerel, yellowfin tuna, skipjack tuna, chub mackerel, and largehead hairtail.

Figure 41. Reported World Catch of Orange Roughy (1970-2005)



Source: Data from the FAO Fishery Statistics Database (FAO 2007b).

While falling catches are not direct evidence of the status of the stock, in a report prepared for the FAO Maguire *et al.* (2006) presented a summary status report:

- SW Pacific – orange roughy are fully to over-exploited;
- SE Atlantic – status of orange roughy is unknown; and,
- NE Atlantic – orange roughy status unknown to fully-exploited.

They concluded that orange roughy should be regarded as overexploited or depleted in all areas where fishing has developed.

The case of the cod fishery in Eastern Canada is also of interest. Traditionally, Northern cod has been caught close to the inshore in the summer, and over the century prior to the 1950s, total catches averaged around 200,000 tonnes/year – from a resource that was clearly one of the most productive and valuable in the world. Since the 1950s, cod began to be harvested in the winter, and in areas further offshore. Total catch peaked in 1968 at over 800,000 tonnes (85% caught by foreign vessels offshore), far in excess of estimated biomass growth (Grafton *et al*. 2000).

As a result of such heavy fishing pressure, and despite later recovery efforts, the stocks continued to decline.[88] By 1991, the total catch was 171000 tonnes (less than the TAC) (Grafton *et al*., 2000); and by 1993, the six major stocks in Eastern Canada had collapsed to the point where a complete fishing moratorium was declared. Initially, the stocks were expected to recover in 3-4 years after the closure, but to date, many stocks have still not recovered[89] and the overall biomass continues to be very low (WWF, 2007).

The cause of this collapse has been a subject of vigorous debate. On the one hand, environmental factors (including climate change, seal predation, or changes in the ecosystem) have been blamed (*e.g.* Lear and

88 Concerns over the socio-economic impact of reduced harvests on the industry prevented decision-makers from reducing fishing pressure. Instead, a "50% rule", which limited reductions in the TAC from year to year, was instituted (Grafton *et al*., 2000).

89 It has been suggested that the divergence between the original predictions and the actual outcome may be attributed to the fact that the predictions were based on the reproductive potential of the stocks when they were in a healthier, more fertile, state (http://www.ices.dk/marineworld/recoveryplans.asp).

Parsons, 1993; Mann and Drinkwater, 1994). While ecological variation may indeed have played a role, there is empirical evidence suggesting that over-fishing was also a contributing factor (*e.g.* Hutchings and Myers, 1994; Myers and Cadigan, 1995a,b; Myers *et al.*, 1996, 1997). For example, Myers *et al.* (1996) analysed historic tagging data, to reconstruct fishing mortality of three of the Newfoundland stocks. They found evidence of "very high rates of exploitation in the late 1980s and early 1990s that are consistent with the hypothesis that these populations collapsed because of over-fishing".

It has been suggested that there are similarities between the fate of cod stocks in Eastern Canada and cod stock development in the North Sea, indicating that *North Sea cod* may also be nearing collapse (see MacGarvin, 2001; WWF, 2007). The estimated stock of North Sea cod is about 53,000 tonnes, which is only one-third of the 150,000 tonnes that scientists recommend as a bare minimum.[90] According to ICES data, there are a number of other North Sea fish stocks whose spawning stock biomass has been identified to be at reduced reproductive capacity (including cod, sandeel, and Norway Pout), or where fishing mortality indicates unsustainable harvesting of the stock (including cod) (Table 54).

Table 54. Assessment of Fisheries in the North Sea Eco-region*

Stock	State of stock		
	Spawning biomass in relation to precautionary limits	Fishing mortality in relation to precautionary limits	Fishing mortality in relation to high long-term yield
Cod (North Sea, Eastern Channel, Skagerrak)	Reduced reproductive capacity	Harvested unsustainably	Overexploited
Cod (Kattegat)	Reduced reproductive capacity	Uncertain	Uncertain
Haddock	Full reproductive capacity	Harvested sustainably	Overexploited
Whiting	Unknown	Unknown	Unknown
Saithe	Full reproductive capacity	Harvested sustainably	Appropriate
Anglerfish	Unknown	Unknown	Unknown
Plaice (North Sea)	At risk of reduced reproductive capacity	Harvested sustainably	Overexploited
Plaice (Eastern Channel)	Unknown	Unknown	Unknown
Plaice (Skagerrak and Kattegat)	Unknown	Unknown	Unknown
Sole (Skagerrak and Kattegat)	Full reproductive capacity	Unknown	Unknown
Sole (North Sea)	At risk of reduced reproductive capacity	At risk of being harvested unsustainably	Overexploited
Sole (Eastern Channel)	Full reproductive capacity	Harvested sustainably	Overexploited
Sandeel	Reduced reproductive capacity	-	Unknown
Norway pout	Reduced reproductive capacity	-	-
Herring (autumn spawning)	Full reproductive capacity	At risk of being harvested unsustainably	Overexploited
Herring (spring spawning)	Unknown	Unknown	Unknown
Sprat	Unknown	Unknown	Unknown
Mackerel	Unknown	Unknown	Unknown
Horse mackerel	Unknown	Unknown	Unknown
Rays and skates	Unknown	Unknown	Unknown

Note: * The North Sea eco-region includes the North Sea, Skagerrak, Kattegat, and the Eastern Channel.
Source: ICES (2006).

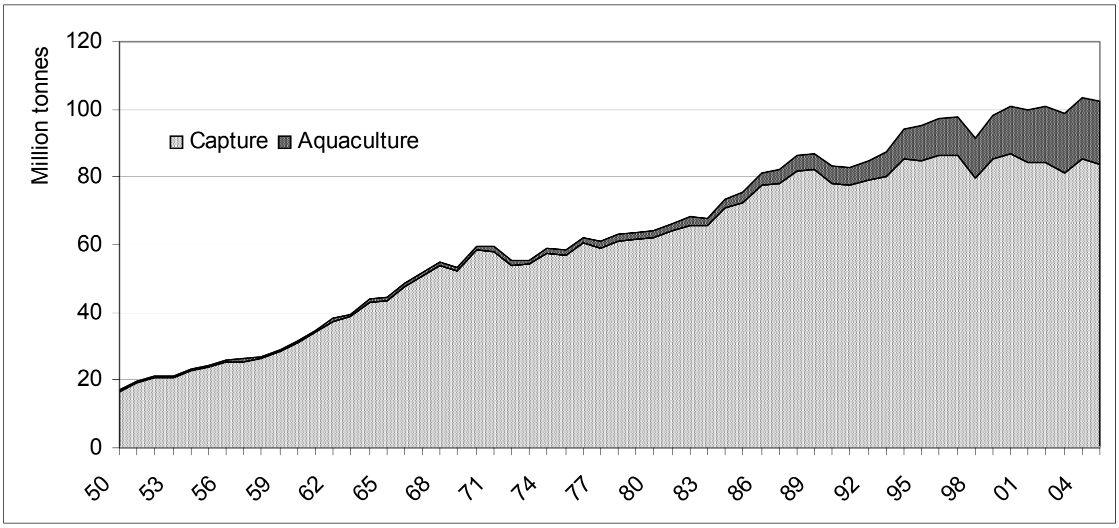
Trends in world marine capture production

World fishery production in 2004 – the total of marine and inland capture and aquaculture production – reached a new high of 140.5 million tonnes, of which 95 million tonnes (68%) was from capture fisheries, and 45.5 million tonnes (32%) was supplied by aquaculture (Figures 42 and 43). While the supply of fish

90 http://www.ices.dk/marineworld/recoveryplans.asp.

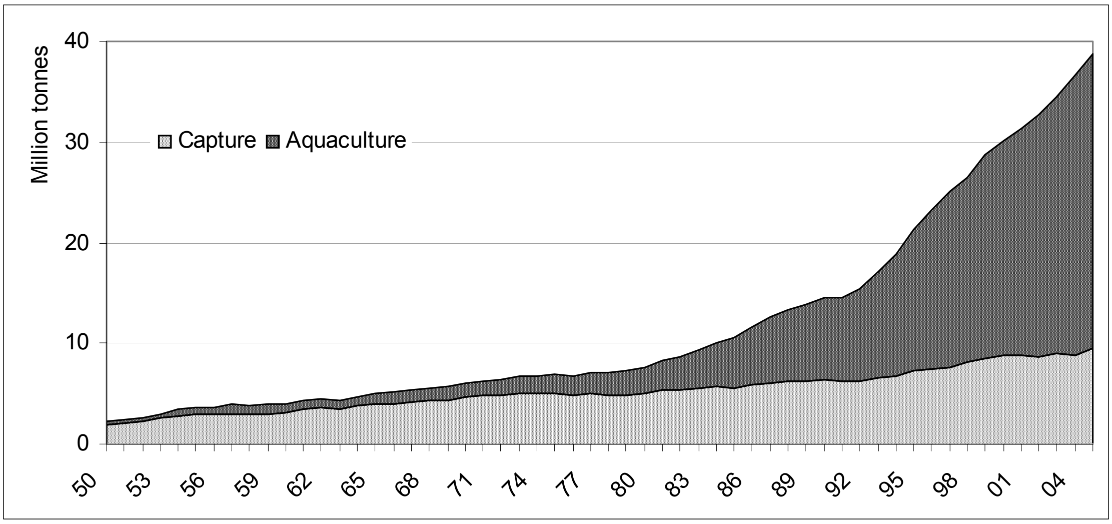
from capture fisheries has been flat in recent years, world aquaculture production has increased by 28% since 2000 (largely due to a rapid growth of aquaculture in China).

Figure 42. World Capture and Aquaculture Production from Marine Fisheries (1950-2005)



Note: Production excluding aquatic plants.
Source: Data from the FAO Fishery Statistics Database (FAO, 2007b).

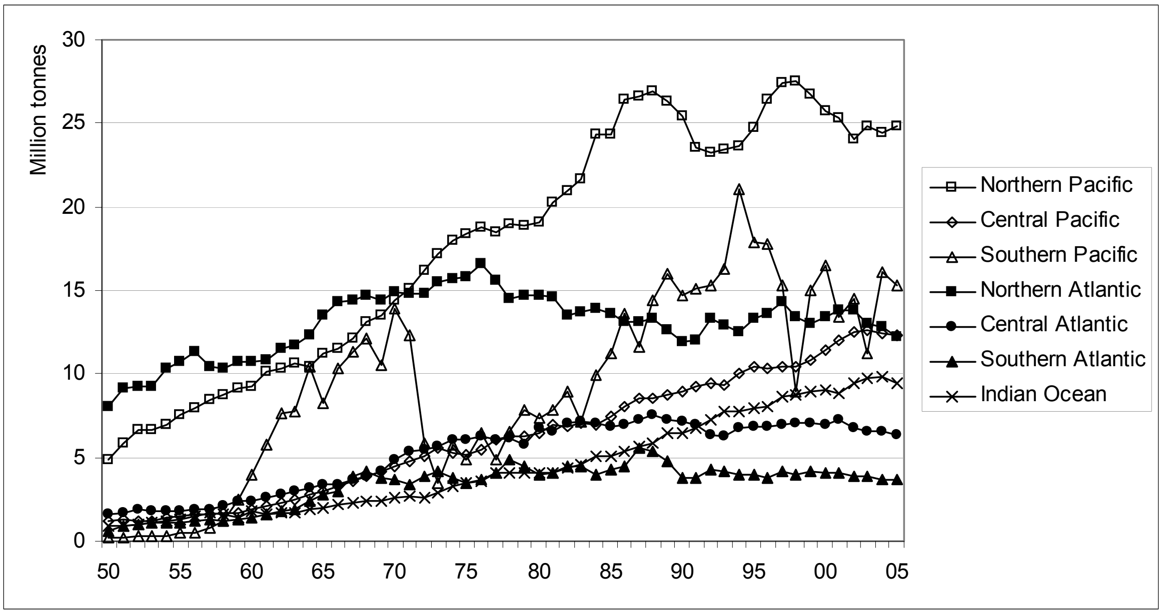
Figure 43. World Capture and Aquaculture Production from Inland Fisheries (1950-2005)



Note: Production excluding aquatic plants.
Source: Data from the FAO Fishery Statistics Database (FAO, 2007b).

By the early 2000s, all the world's major production regions had reached their peak, and few regions now have a substantial number of under-exploited or moderately exploited stocks leaving little room for further expansion (FAO, 2007a) (Figure 44). The world marine capture production may have reached its maximum potential under the current management framework.

Figure 44. Capture Fisheries Production in World Oceans



Note: Production excluding aquatic plants.
Source: Data from the FAO Fishery Statistics Database (FAO, 2007b).

Reasons for inaction: Why is fisheries management frequently unsustainable?

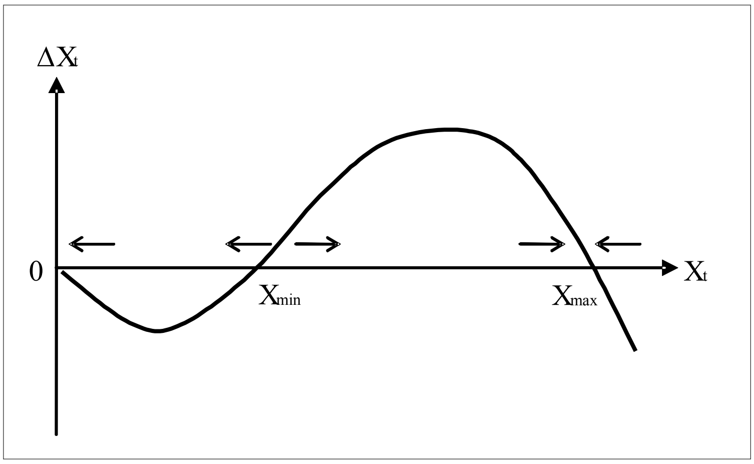
There are a number of factors which contribute to the complexity of fisheries management, including the "common-pool" nature of the resource, a variety of information problems, market imperfections (*e.g.* sticky markets), the presence of uncertainty, as well as the challenge to regulate fishing and enforce fisheries management measures at reasonable administrative cost.

The potential costs are exacerbated by the renewable nature of the resource. In the case of a non-renewable resource, over-exploitation will have only temporal ramifications for the potential harvest of the resource (*i.e.* the resource will be extracted at a slower (or faster) rate than is economically optimal). However, consequences of over-exploiting a renewable resource may be much more severe: over-exploitation may not only temporarily reduce the harvest, it may also steer the resource onto an irreversible path toward commercial extinction.

Management of fisheries is complex

Understanding the population dynamics of fisheries is a key to their sustainable management. Figure 45 illustrates a fishery production function which approximates the biological growth dynamics of the resource. The stock of fish biomass at time t (denoted as X_t, and shown along the horizontal axis) determines the resource biomass in the following time period (X_{t+1}). Biomass growth is then expressed as the difference between the two stocks, $X_{t+1} - X_t$ (denoted as ΔX_t, and shown along the vertical axis). In addition to the point of origin (where the stock equals zero), two fixed points characterize the population dynamics of the resource: a *maximum carrying capacity* (X_{max}) -- *i.e.* the biomass which can be sustained in a given ecosystem in the long-run. Conversely, a *minimum viable population* (X_{min}) is necessary for the fish to attain positive growth rates. Sustaining the stock of fish above the minimum level (X_{min}) is critical for fish survival. When the stock drops below X_{min},, the species is generally on a path headed for extinction.

Figure 45. Example of a Biomass Growth Function



Therefore, reduced yields today may allow for increased yields in the future – as the stock rebuilds itself. As such, the critical question is how much of the resource should be harvested at any given point in time? This requires an understanding not only of biomass growth, but also of the costs and efficiency of effort, and the returns on fish in commercial markets. The maximum sustainable yield is likely to be greater than maximum economic yield, since the latter takes the cost of effort into account.

Competitive markets will generally deliver this efficient outcome if the underlying property rights are well-defined. However, marine fisheries have the characteristics of a "common-pool" resource, and are often plagued with market imperfections.[91] Hence, it is unlikely that socially-optimal harvest rates can be sustained without some form of government intervention. Under open-access conditions, a vessel will continue fishing until the marginal costs of harvesting an extra fish equal the marginal revenue from selling the fish -- not taking into account the fact that every additional fish harvested decreases the overall stock of fish, and thus contributes to decreasing biomass growth rates in the future. In other words, open-access conditions in a "common pool" fishery create a situation where the bio-economic externality is not internalised by the fishing industry. In such conditions, conservation is "an unlikely and an unstable outcome" (Conrad, 1999). A non-cooperative equilibrium results instead, with the individual fishers behaving as if "there were no tomorrow", by employing an infinite discount rate to evaluate the benefits of conservation" (Conrad, 1999). Therefore, public policy intervention is needed if fisheries are to be exploited in an efficient and sustainable manner.

Imperfect information and conflicting policy objectives

In practice, regulation of fisheries is complicated by: (i) uncertainty concerning the status of fish resources; and (ii) involvement of stakeholders who have conflicting policy objectives. Uncertainty about the status and dynamics of fish stocks may be considerable. For example, in the North Sea eco-region, the status of 8 species items (out of 27 in total) is uncertain or only partially known. The status of a further 16 species items is reported as "unknown" (ICES 2006: 19-21). This uncertainty complicates the task of fishery biologists when providing scientific advice for the management of fisheries.

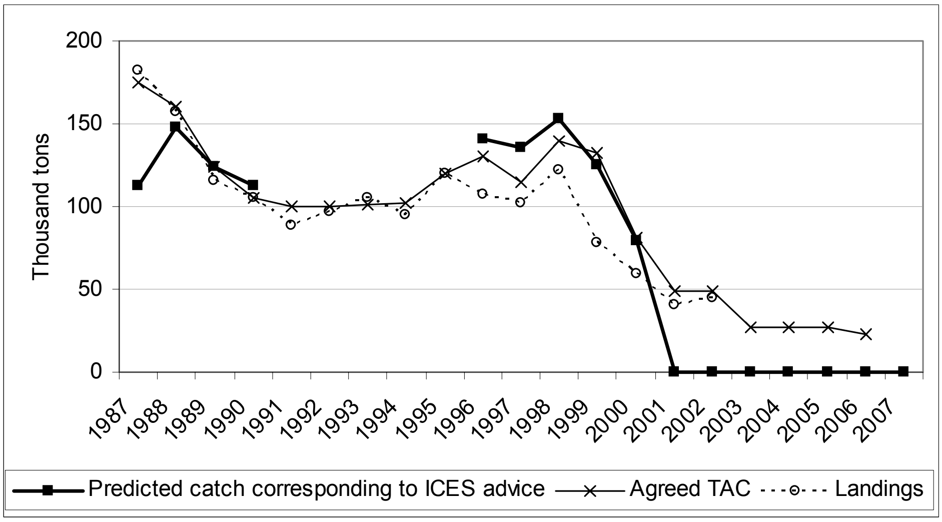
Another factor is that stakeholders (*e.g.* labour unions, industry representatives, and environmental groups) other than fishery biologists also take part in the decision-making process. Ideally, reconciliation of the potentially conflicting objectives would result in management decisions which balance the socio-economic

91 In effect, the resource is not recognised as "private property" until it is captured. For more on "common-pool resources" see (for example) Hartwick and Olewiler (1998); Conrad (1999).

goals with resource sustainability. However, there is evidence indicating that past management decisions may have leaned towards socio-economic considerations, to an extent which is incompatible with resource sustainability.

For example, in Europe, guidance related to the management of marine fisheries is issued by the Advisory Committee on Fishery Management (ACFM) -- one of the committees of the International Council for the Exploration of the Sea (ICES). The scientific advice is then taken into account by fishery policy-makers when determining quota and other restrictions imposed on a fishery. In a recent report, ICES classified the stock of cod in the North Sea eco-region (including the North Sea, Eastern Channel, Skagerrak, and Kattegat) as being harvested unsustainably, and suffering reduced reproductive capacity. ICES has therefore recommended a zero-catch, until initial recovery of the cod spawning stock biomass has been proven (ICES, 2006). However, despite ICES's advisory to close the fishery, every year since 2001, fishery policy-makers have continued to set non-zero TACs [92] in all sub-regions of the North Sea eco-region (Figures 46-48). Economic concerns have likely played a role in these decisions.

Figure 46. Advice, Allowable Catch, and Actual Landings: Cod in the North Sea



Economic considerations may have also played an important role in the decision to close the fishery of Norway pout during 2005-2006. Norway Pout is an important prey species for a variety of predator species (cod, whiting, and saithe). It is harvested as an industrial species for the production of fish meal. Lowering the catch of this stock increases the availability of the prey for predator fish, including North Sea cod.

Norway pout is generally considered to be a very resilient species. According to ICES, the population dynamics of Norway Pout in the North Sea are very dependent on changes caused by variation in recruitment rates,[93] in predator mortality, or other natural mortality causes. Recruitment of Norway pout is highly variable, and this variability influences the stock of spawning biomass rapidly, due to the short life span of the species (ICES, 2006).

92 TAC = Total Allowable Catch, which refers to the maximum volume of harvest imposed as a management rule.

93 "Recruitment" refers to the net increase in biomass due to maturing of younger age classes and, depending on species, maybe due to in-migration of adult fish from other populations.

Figure 47. Cod in Skagerrak

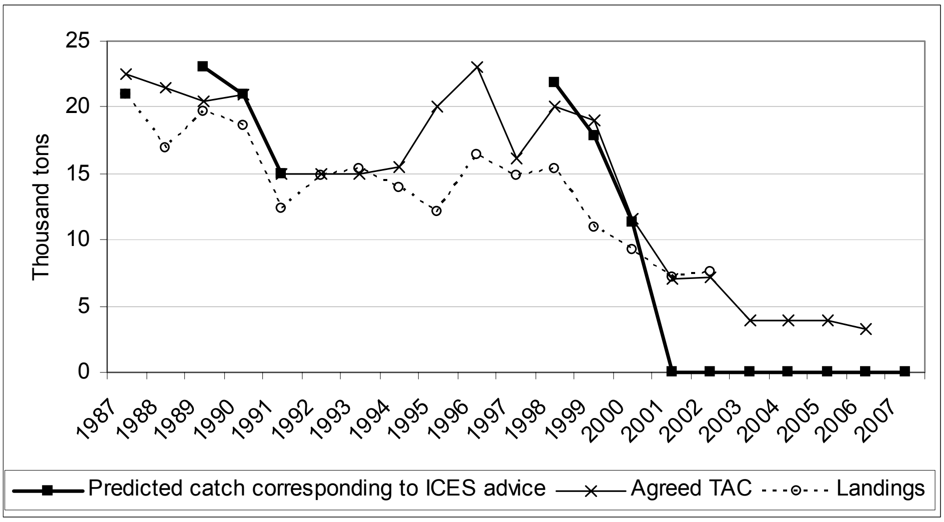
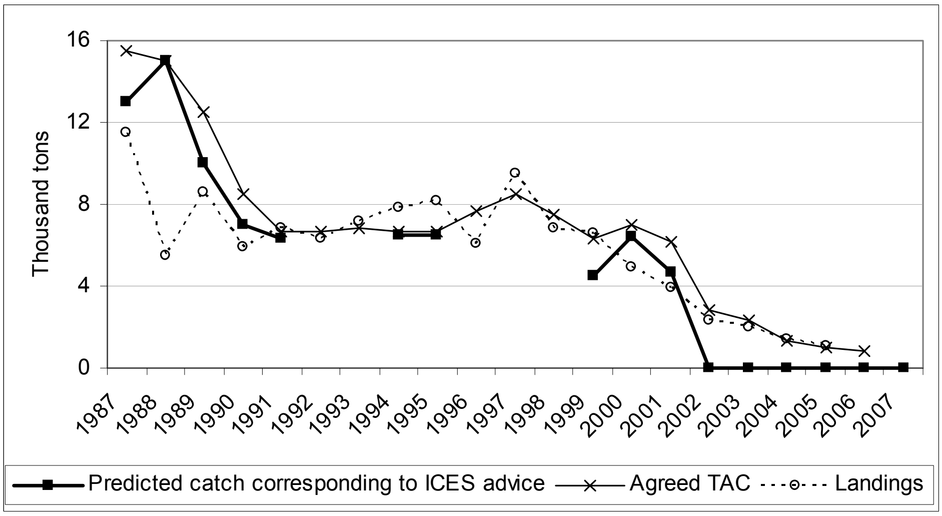
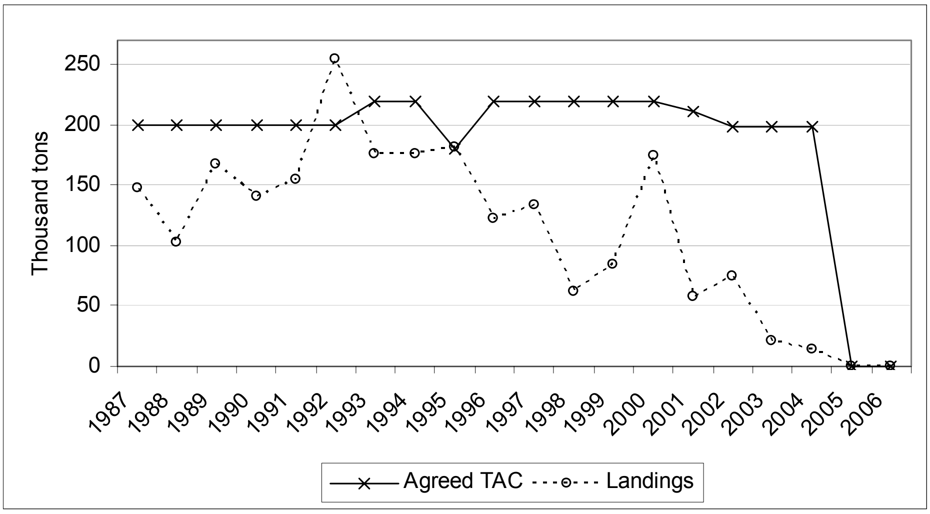


Figure 48. Cod in the Kattegat



Source: Data from ICES (2006).

Figure 49. Allowable Catch and Actual Landings: Norway Pout in the North Sea

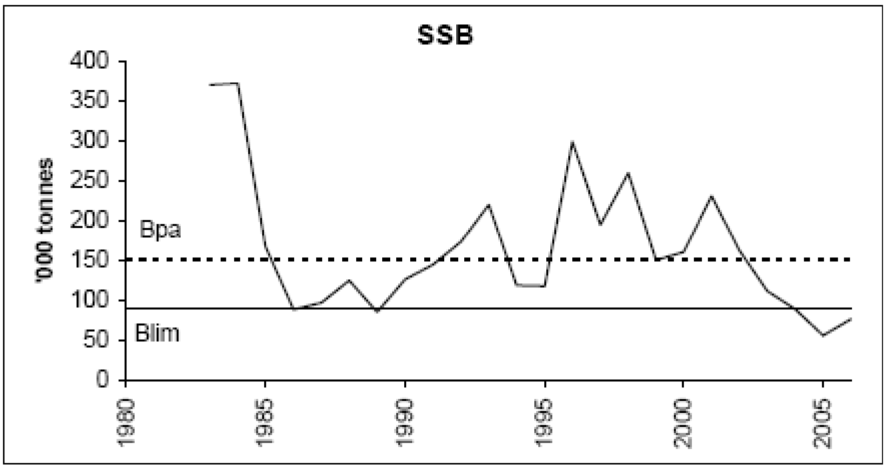


Note: Landings = Actual volume of catch, estimated by adjusting reported official landings for non-reporting, misreporting by fishing area, and discards (referred to as 'ACFM landings' in ICES 2006).
Source: Data from ICES (2006).

The expectation that the level of the TAC has little effect on the stock of this highly resilient species is evidenced in Figure 49. While the level of TAC for Norway pout has been more-less constant, there has been large variation in the level of actual landings – well below the permitted quota during most of the 1987-2004 period.

However, the sudden drop in spawning stock biomass below safe levels triggered the closure of the fishery in 2005–2006. According to ICES, variation in ecological conditions, rather than unsustainable TAC, was responsible for much of the dramatic decline in the stocks of Norway pout (ICES, 2006). However, the example illustrates the level of uncertainty involved in fisheries management, and the degree of precaution which needs to be exercised. Thus, Figure 50 shows the drop in stock of spawning biomass below a "safe level" (Blim), due to very low recruitment levels in 2003–2004.

Figure 50. Stock of Spawning Biomass: Norway Pout in the North Sea



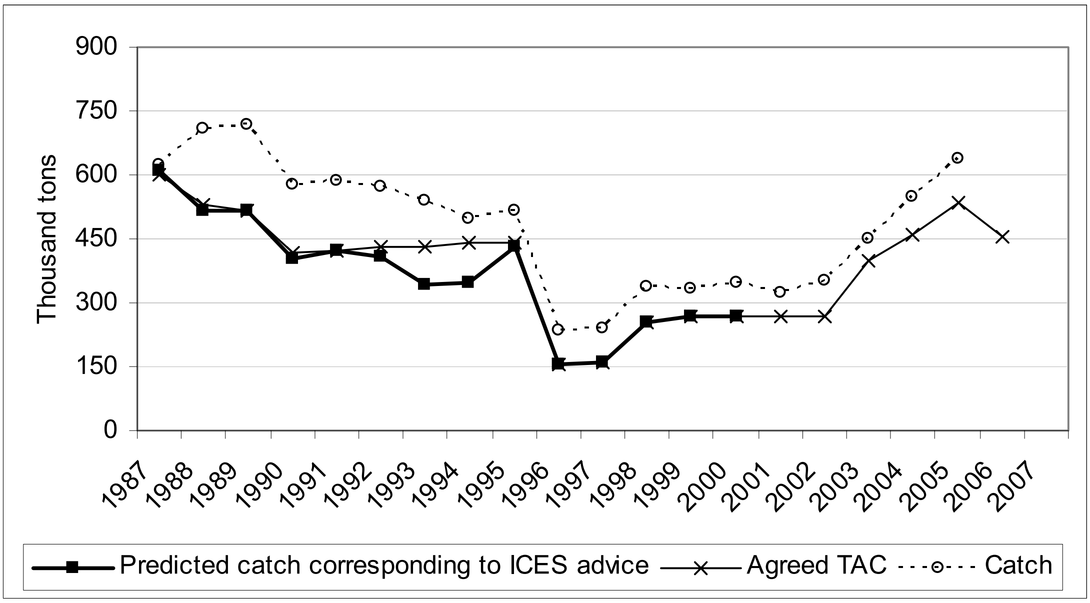
Note: SSB = spawning stock biomass; B_{lim} = limit reference point for SSB defined as stock size with risk of impaired recruitment; B_{pa} = a precautionary reference point for SSB defined as a safety threshold to prevent stocks at limit values.
Source: ICES (2006).

Non-compliance and insufficient policy enforcement

In addition to the lack of reliable data and conflicting policy objectives in the area of fisheries management, inaction with respect to policy enforcement may also result in unsustainable fishery management and over-fishing. The "common-pool" nature of fishery resources, the presence of information asymmetries, as well as agency problems may all cause non-compliance with existing regulations (including quota restrictions). These problems are particularly acute for high seas, straddling and highly migratory species (see Maguire *et al.*, 2006).

However, they also affect fish stocks within defined EEZs. Due to a lack of resources, this is particularly true of developing country EEZs, some of which are exploited by distant water fishing fleets. However, it is also a problem in OECD waters. For example, in the case of the North Sea herring fishery the recommended catch and the TAC were very close, but the regulator has consistently failed to enforce the quota restrictions set out in the total allowable catch (TAC). Non-compliance led to continued over-exploitation of the stock (Figure 51). As a result, the 2006 ICES advisory has recommended reduction in herring catches.

Figure 51. Advice, Allowable Catch, and Actual Catch: Herring in the North Sea and Eastern Channel



Note*:* Data on catch include unallocated and misreported landings, discards, and slipping.
Source: Data from ICES (2006).

Sustainable management in the face of mixed fish stocks

In practice, multiple fish species are often fished at the same time. This complicates fisheries management, and policies that do not sufficiently account for these factors may result in over-fishing. For instance, mixed fisheries are common, with many stocks being exploited together, due to their proximity. Exploitation of mixed fisheries can result in significant by-catch, which may threaten the survival of non-targeted species. Therefore, management must consider the state of individual stocks and their simultaneous exploitation in mixed fisheries, as well as the constraint that harvest technologies cannot perfectly distinguish among species.

Costs of policy inaction with respect to unsustainable fishery management

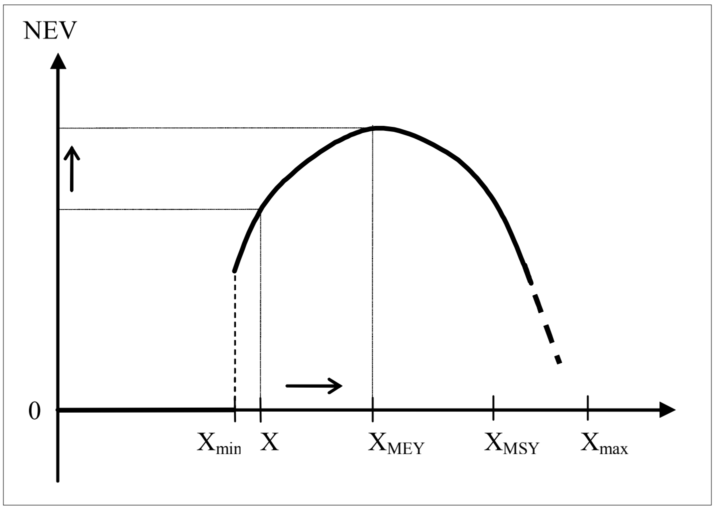
The costs of unsustainable fisheries management are primarily reflected in the loss of commercial yields of the fish stock. However, there may also be ancillary impacts, such as lost employment opportunities in isolated communities with important labour market rigidities. Other impacts such as reduced food security may be significant as well. And finally, the impacts on marine biodiversity can be important.

Foregone income from fisheries

The costs of policy inaction include costs directly associated with stock depletion, including lost future receipts of fishers and vessel owners. The renewable nature of fishery resources means that if a resource is being over-harvested (hence, exploited unsustainably), the associated cost equals the discounted value of the foregone future benefit stream from the resource – until the stock recovers to its full productive capacity. If over-harvesting results in commercial extinction of the stock, the associated cost will be equal to the discounted value of future benefit stream from now forever.

Figure 52 illustrates the discounted net economic value (NEV) of fishing as a step-function, with a discontinuity at the minimum viable biomass (X_{min}). Positive values of NEV are attained as long as stock biomass is maintained above X_{min}. However, NEV is zero if stock biomass drops below this level (*i.e.* commercial extinction of the species is irreversible). If the current stock of biomass is X, this might be considered too low because: (i) it may be too close to the minimum viable biomass, and therefore care should be exercised because unforeseen ecological variability may result in stock collapse (hence, zero NEV); and (ii) an increase in the stock of biomass will bring about higher economic benefits in the future (a move from X toward X_{MEY}). In sum, reduction in current fishing pressure would help the resource recover to its full productive capacity and allow for a higher stock of biomass (well above the risky minimum level), as well as increased exploitation rates in the future.

Figure 52. Potential for Welfare Improvements

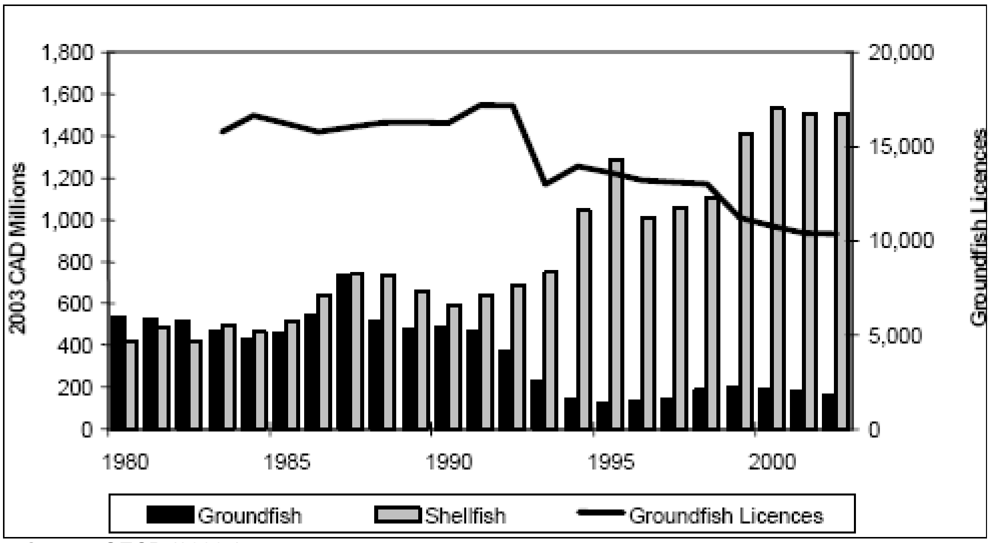


There is empirical evidence suggesting that the value of the lost social welfare associated with unsustainable fisheries management is not negligible. Based on a study of 13 “overfished” fish stocks in US waters, Sumaila and Suatoni (2006) assessed the lost direct use values (commercial fishery yields and recreational fishing) associated with continued excessive fishing effort and found that the lost NPV associated with continuation of the existing management regime was USD 373 million (USD 193.7 million, instead of USD 566.7 million).

Bjørndal and Brasao (2005) estimated the *costs* (in terms of lost net present value) associated with unsustainable management of the East Atlantic Bluefin Tuna fisheries. Due to its migratory nature, the resource is effectively “open access”. Moreover, it is very valuable -- the price for a single fish can be $100,000. The incentives for unsustainable exploitation by fishers are therefore very high. Inappropriate gear selection and excessive effort are both leading to overfishing. Indeed, it has been forecast that if the present situation continues a complete stock collapse is likely. Bjørndal and Brasao (2005) also estimated that the *benefits* from a adopting a more sustainable fisheries management regime with enforced limits on the TAC and restrictions of gear choice would result in an increase in the discounted total net present value of the fishery from $937 million to over $3 billion. Of course, the adoption of such a regime would depend on international co-operation.

In Canada, the closing of the Atlantic cod fishery in 1992 also had important economic impacts. Foregone income from the Atlantic groundfish fisheries directly attributable to the stock collapse reached an estimated CAD 250 million in the short-term (the landed value of the Atlantic groundfish declined from about CAD 400m in the early 1990s, to about CAD 150m in the mid-1990s). In the long-term, the forgone potential annual income from a sustainable fishery reached a substantially higher figure - an estimated CAD 1 billion per year (MacGarvin, 2001). In reality, this estimate was mitigated by the latent potential of shellfish fisheries which had not been exploited previously (MacGarvin, 2001). As a result of these market adjustments, the total value of processed fin and shellfish actually increased (Figure 53).

Figure 53. Landed Value and Number of Groundfish Licences for Atlantic Groundfish and Shellfish (Canada)



Source: OECD (2006c).

Lost employment and increased government assistance

The *indirect* costs of over-fishing may also be substantial. Fisheries are an important source of employment in fish-processing industry and related sectors. In the short-run, collapse of fisheries may cause loss of local employment. Some of these costs are, of course, reflected in the loss of the commercial yield of the fishery. However, they may not be fully reflected in the value of the lost yield. For instance, although markets may be able to absorb some employment losses in the long-run (for example, through regional labour migration or changes in sectoral structure), substantial adjustment costs may also be associated with this process. Such costs are magnified in the case of fisheries, since labour markets tend to be rather “sticky”, largely due to the skewed age structure of workers (a relatively elderly labour force), their education and qualification profile (low educational attainment, sector-specific skills), and socio-economic and cultural aspects of fishing communities (cultural affinity to the sector, distance from other labour markets). In the face of such adjustment costs, additional public funds may need to be disbursed in the form of employment assistance programmes and other forms of government aid (OECD, 2000).

Paradoxically, in some countries, unsustainable government policies contributed to excess capacity in the fishery sector and exacerbated subsequent problems of exit (overcapitalization), labour migration, and social dislocation, as well as depletion or collapse of fish stocks (with ensuing impacts on marine ecosystems). For example, in Korea, negative labour market impacts (triggered by unsustainable policies) emerged in the early 1990s. According to OECD (2000), past government support policies encouraged the expansion of fishing capacities, and thus contributed to over-fishing in Korea's marine waters. The resulting decline in fishing profitability discouraged young workers (age 15-29) from entering the sector and contributed to their out-migration from fishing communities as they sought alternative employment ("young" employment in fishing fell by 80% between 1981 and 1994). Training programmes and recruitment of foreign workers were then introduced, in an attempt to alleviate the shortages.

In Canada, the closing of the Atlantic cod fishery in 1992 had important consequences for employment. Until 1992, groundfish (especially cod) had been a key foundation of the local economy. In the Province of Newfoundland and Labrador, groundfish supplied approximately 80% of total revenue (in some communities, this reliance was effectively 100%), and one person in five was employed in the fishery. In 1990, more than 800 fish plants employed 60,000 workers, and 26,000 families depended on fish-processing to earn living (OECD, 2006c). It has been estimated that about 30,000 people lost their jobs at the height of the crisis, including 10,000 fishermen (about 25% of all tax-filing fishers[94]) (MacGarvin, 2001).

The negative impacts on the labour market were, to some degree, mitigated through subsequent market adjustments, including: (i) expansion of the shellfish fishery; (ii) the switch to imported fish by the processing industry (MacGarvin, 2001); and (iii) inter-provincial migration. For example, it has been reported that 14,500 people (tax-filers) migrated from Newfoundland to other Canadian Provinces. When factors such as changes in incomes and unemployment rates are controlled for, the net outflow has been estimated at between 18,000 and 26,000 people (Day and Winer, 2001).

In response to the crisis, substantial public funds were spent on income support (including fishers' unemployment benefits) and government assistance programmes (expenditures towards restructuring, sectoral adjustment, and regional economic development). Three of the government assistance programmes contained a license retirement component (Table 55). In total, CAD 3.5 billion was spent on income support, industry adjustment measures and economic development assistance programmes (OECD, 2006c).

The total federal government expenditure on fisheries grew from about CAD 150 million per year in the mid-1980s, to some CAD 700 million per year in the mid-1990s (most of this was attributed to the closure of cod fishery). This amounted to an annual expenditure of about CAD 6500-9000 per affected individual (Table 56).

[94] This includes both Atlantic and Pacific coast fishers – where a similar collapse of Pacific salmon fishery occurred.

Table 55. Assistance Programmes for the Atlantic Fishery with a License Retirement Component (1992-2001) (CAD million)

	Program Components / Year			
	NCARP 1992-1994	TAGS 1994-1998	CFAR 1998-2001	Total
Income Replacement	484	1,750*	315	2,549
Training & Counselling	333		0	333
Vessel Support Program	15	12	0	27
Early Retirement	31	28	85	145
Licence Retirement	40	60	230	330
Economic Development	0	50	100	150
TOTAL	903	1,900	730	3,533

Note: * Includes money for Training & Counselling; NCARP = Northern Cod Adjustment and Recovery Programme; TAGS = The Atlantic Groundfish Strategy; CFAR = Canadian Fisheries Adjustment and Restructuring Plan.
Source: OECD (2006c).

Table 56. Selected Income Support and Special Adjustment Programmes Aimed at Atlantic Fisheries in Canada

Programme	Total budget (in Canadian dollars)
Atlantic Fisheries Adjustment Programme, Quebec Federal Fisheries Development Programme	637 million
Northern Cod Adjustment and Recovery Programme (NCARP)	587 million
Atlantic Groundfish Adjustment Programme	381 million
The Atlantic Groundfish Strategy (TAGS)	1 900 million

Source: MacGarvin (2001).

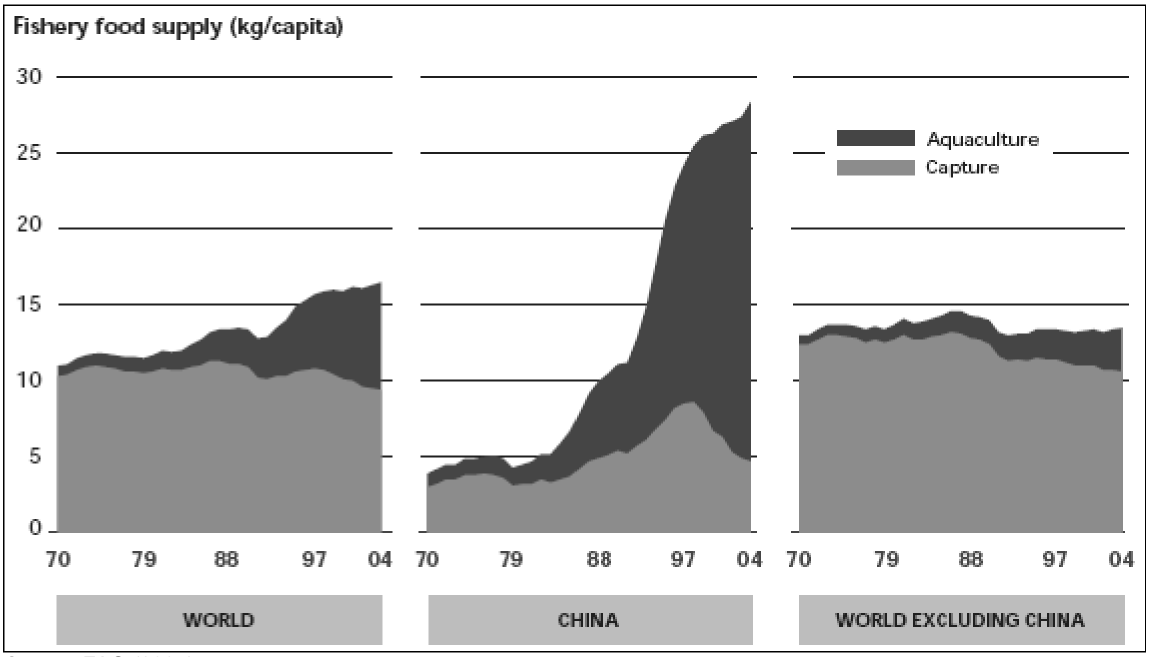
The combination of forgone income and government spending amounted to an estimated total of CAD 1.75 billion annually (MacGarvin, 2001). However, the fishery sector is plagued with many market imperfections, so obtaining credible estimates of social costs using market prices is virtually impossible. In addition, the social costs associated with the stock collapse are likely to be substantially higher than this, for several reasons. First, this estimate does not include the "deadweight cost" associated with collecting and redistributing government funds. Second, lost recreational opportunities were not taken into account. Third, no consideration was made of the value of lost marine ecosystem services.

Threatened food supply and food security

Fish are an important source of animal protein for human consumption, and a key source of nutrition in many remote coastal areas. Globally, about 17% of the animal protein supply is derived from fisheries. In many developing countries -- especially in Asia -- this share is above 50% (FAO, 2005). In many countries, unsustainable management of fisheries may thus have negative impacts on food supply and food security. Depending on the availability of food substitutes, the associated costs may include losses in consumer welfare, as a result of changes in the composition of daily food intake due to: (i) increased price of fish (income effect); and (ii) the necessity to compensate for changes in the nutritional value of the new food items (substitution effect). Public funds may also be needed to mitigate the negative impacts of reduced availability of fish for food. In the case of the most fish-dependent communities, provision of emergency aid may be required in the face of stock collapse.

Although cost estimates associated with policy inaction with respect to food supply and food security are rarely available, there is some evidence indicating that significant structural changes in the supply of fish for human consumption have occurred recently. At the global scale, rising population levels place an increasing demand on supply of fish for human consumption. Since the potential of marine capture fisheries is limited, demand for fish has increasingly been met by aquaculture production. This trend is evident from Figure 54, which shows that while capture production has remained more-less constant in past decades, and has even started to decline. However, the supply of fish from aquaculture has been growing -- in both absolute and relative terms -- as a proportion of total fish consumption.

Figure 54. Relative Contribution of Aquaculture and Capture Fisheries to Foodfish Consumption



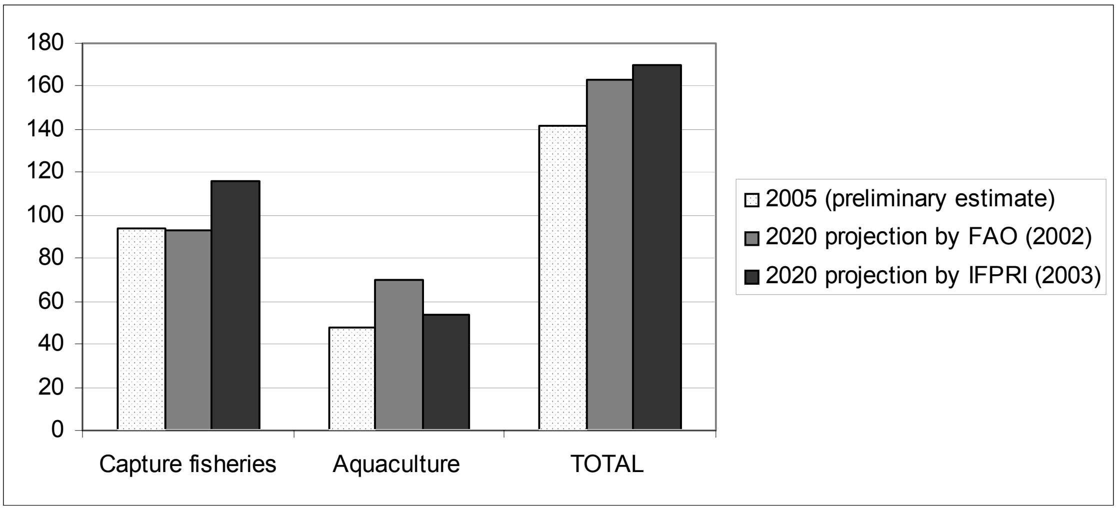
Source: FAO (2007).

Aquaculture has been growing more rapidly than any other animal food-producing sector[95] -- at an average annual rate of 8.8% per year since 1970, compared with only 1.2% for capture fisheries, and 2.8% for terrestrial farmed meat production systems (FAO, 2007a).[96] Although total supplies of fish for food have been stagnating in recent years, per capita supplies from aquaculture have increased strongly. This is particularly so in China, where supplies from aquaculture provide about 83% of total food fish supplies, compared to only 21% in the rest of the world (FAO, 2007a). Overall, fishery production is projected to increase further by 2020. However, available studies disagree on the relative importance of capture fisheries versus aquaculture in contributing to the 16-21% projected growth (Figure 55).

95 Mostly freshwater culture, followed by mariculture and brackish-water culture (FAO, 2007).

96 Of the world total, China is reported to have accounted for nearly 70% of aquaculture production (in quantity terms). However, FAO has expressed some reservations about the reliability of data on China's aquaculture production, suggesting that they are likely too high (FAO, 2007).

Figure 55. World Fisheries and Aquaculture Production (Million Tonnes)



Note*:* Production excluding aquatic plants.
Source: FAO (2007).

Disruption of local livelihoods and efforts for economic development and poverty alleviation

Fish stocks are mobile natural resources, which do not respect international boundaries. In the course of a given year, a fish stock may migrate from the open ocean, far off the coast to inshore waters, or *vice versa*. Offshore fishery exploitation will necessarily impact the size of the stocks when they migrate inshore. Hence, over-exploitation of fisheries at the global scale (outside the 200-mile zone) may disrupt and threaten local livelihoods, including coastal fishery-dependent communities and other small-scale (subsistence and artisanal) fisheries.

Although the contribution of the fishery sector to national economic output is typically relatively low (about 0.5-2.5%), its economic importance for individual coastal communities may be very high. In many developing countries, for example, small-scale fisheries are an important means of livelihood. Ninety per cent of the 38 million fishers and fish-farmers world-wide in 2002 were classified as "small-scale". The contribution to economic development can be significant. Developing countries' share of world fish exports rose from 40% in 1980, to 50% in 2001, with net receipts from fish trade increasing during the same period from $4 billion to $18 billion (FAO, 2005). The income multiplier effect has also the potential to contribute to economic growth, and can also be an important source of government tax receipts (FAO, 2005).

Since fishery activities are often carried out in combination with agricultural or forestry activities, they may also help to "insure" households against production risks in any one of these other activities. There are also important intra-sectoral interactions (*e.g.* between capture fisheries and aquaculture, or agriculture and aquaculture, through the supply of fishmeal) and inter-sectoral interactions (*e.g.* between forestry and fisheries through the supply of timber for boat-building) (see *e.g.* FAO 2005).

While many developing countries were previously significant net exporters of food, it is expected that they may become net importers of food in the future. For the poorest countries -- most of which are found in sub-Saharan Africa and South Asia -- the financing of food imports will therefore become a high priority, and capture fisheries and aquaculture will come under strong pressure to help out (FAO, 1999). The potential of fisheries to contribute to the reduction of hunger and malnutrition in low-income food-deficit countries has been recognized by FAO's Special Programme for Food Security.[97] Of the 30 countries most dependent on fish as a protein source, all but four are located in the developing world.[98] Unsustainable fisheries resource management will, therefore, hit developing countries particularly hard.

Reduced recreation opportunities

Over-exploitation of commercial fisheries in marine areas will also have important impacts on recreational fishing in coastal, as well as inland areas (anadromous, or migratory, species). Recreational fishing associated with marine fisheries generate important economic benefits through their impact on the local economy (*e.g.* tourism revenues), as well as benefits that recreational fishers may experience directly themselves. For example, the value of leisure and recreational benefits derived from marine environments in the UK has been estimated at 11.77 billion GBP per year, using market prices (Beaumont *et al.*, 2006).

In addition to the benefits which result from market transactions, there are also other benefits of marine recreational fishing which are not fully reflected in market prices. Contingent valuation and travel cost methods have been used to estimate the value of recreational fishing in these contexts. For example, Paulrud (2006) estimated the marginal willingness-to-pay to improve sport-fishing conditions in southern Sweden. For coastal angling, they estimated the value of an extra fish caught at $0.56 ($1.33 per kg). In another study, Wheeler and Damania (2001) estimated the marginal value for extra catch for coastal angling off the coast of New Zealand at US$0.67-9.44 per fish (US$1.11-2.78 per kg). Many studies have also attempted to estimate the value of recreational fishing opportunities in the US and Canada (Table 57 provides a partial list).

Chavez-Comparan and Fisher (2001) estimated the consumer surplus of recreational fisheries based on fishing trips to Manzanillo, on the Pacific coast of Mexico. The estimated consumer surplus varies between $7.14 to $39.10 per fishing day, according to the valuation technique employed. The aggregate consumer surplus of recreational fisheries was calculated by multiplying the value of the estimated consumer surplus per fishing day by the total number of anglers who benefited from the use of the resource. The estimates range between $170,001 to $930,697 per year.

Leon *et al.* (2003) estimated the benefits received by international tourists from big-game fishing in Gran Canaria (Canary Islands, Spain). The mean WTP values are estimated to be EUR 72.85 for the use value of big-game fishing, and EUR 56.87 for the preservation value ("existence" and "option") of the stock biomass. These benefits are significant, relative to the size of the tourism revenues. The sum of the use and preservation values (EUR 129.72) represents 9.18% of the average total expenditure per holiday across the sample. In aggregate terms, they estimated the value at EUR 1.4 million. The total benefit to the local economy resulting from this expenditure will depend on the multiplier effects throughout the different economic activities. In addition, the aggregate consumer surplus from both use and preservation benefits amounts to an estimated EUR 726,432, which represents 51% of the estimated amount of directly-related expenditures.

In the US, coastal and marine recreational fishing attracts millions of participants every year generating substantial tourism revenues (Zhang and Lee, 2007). Fishing-related tourism is a crucial industry for some states and regions and the impact of these activities on the State economies may be significant. For example, in Florida, 3.1 million people participate in fishing recreation, contributing $3.8 billion to the local economy (Zhang and Lee, 2007).

97 http://www.fao.org/focus/e/SpeclPr/SProHm-e.htm.

98 http://www.fao.org/focus/e/fisheries/intro.htm.

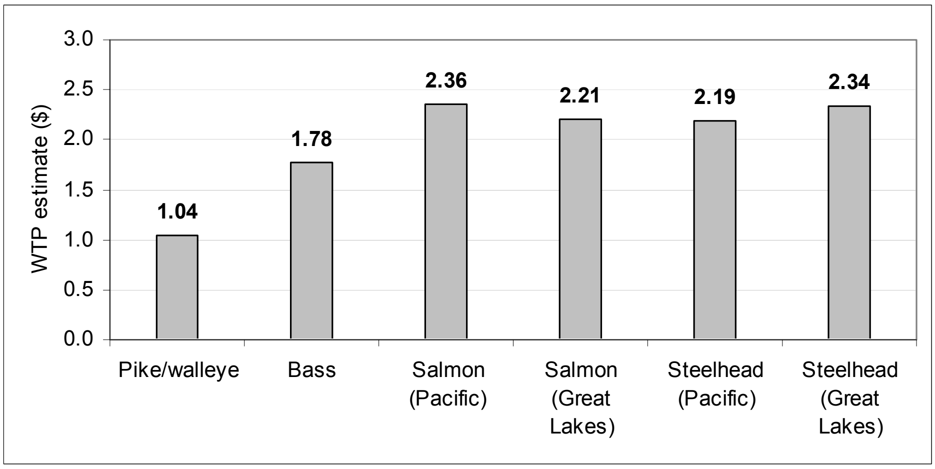
Table 57. Recreational Fishing Valuation Studies for Marine and Anadromous Species (USA and Canada)

Study	Regions	Methodology	Marginal value per fish (in 2003 dollars)
Agnello (1989)	FL-NY	Travel cost	Bluefish ($0.70-9.23) Flounder ($3.33-28.67) Weakfish ($0.05-9.69)
Alexander (1995)	OR	Nested RUM	Steelhead trout ($3.59-23.17)
Berrens *et al.* (1993)	OR	CV (payment card)	Chinook salmon ($3.99)
Bockstael *et al.* (1989)	MD	Travel cost	Striped bass ($2.23)
Cameron and Huppert (1989)	CA	CV (payment card)	Salmon ($5.82-16.76)
Cameron and James (1987a)	BC	CV (dichotomous choice)	Salmon ($2.51)
Cameron and James (1987b)	BC	CV (dichotomous choice)	Salmon ($19.78)
Carson *et al.* (1990)	AK	CV (payment card, conjoint analysis)	Chinook salmon ($15.80-45.92)
Gautam and Steinbeck (1998)	ME, NH, MA, RI, CT	Travel cost; Non-nested RUM	Striped bass ($4.18-7.02)
Hicks (2002)	NH-VA	CV (conjoint analysis); Non-nested RUM	Summer flounder ($2.59-4.65)
Jones and Stokes Associates (1987)	AK	Non-nested RUM	Halibut ($153.91) Chinook salmon ($325.29) Coho salmon ($178.65)
Loomis (1988)	OR, WA	Travel cost	Steelhead trout ($40.69-182.23) Salmon ($13.23-114.21)
Morey *et al.* (1991)	OR	Non-nested RUM	Salmon ($5.66) Ocean perch ($13.74)
Morey *et al.* (1993)	ME	Nested RUM	Salmon ($386.63-612.79)
Norton *et al.* (1983)	ME-NC	Travel cost	Striped bass ($3.39-31.98)
Olsen *et al.* (1991)	WA, OR	CV (open-ended)	Salmon ($21.95-37.44) Steelhead trout ($37.00-81.29)
Rowe *et al.* (1985)	CA, OR, WA	Non-nested RUM	Coastal pelagics ($3.82-4.45) Salmon ($7.21-31.24)
USEPA (2004)	CA	Non-nested RUM	Salmon ($8.46-15.56) Sea Bass ($0.36-0.73) Striped bass ($4.31-8.41)
USEPA (2004)	NY-VA	Nested RUM	Bluefish ($6.32-6.42) Striped bass ($15.52-15.56) Weakfish ($14.31-14.99)

Source: Johnston *et al.* (2006).

Figure 56 illustrates the marginal WTP for recreational fishing, based on a meta-analysis conducted by Johnston *et al.* (2006), using a sample of 48 North American recreational fishing valuation studies. The results suggest a high degree of homogeneity in preferences. For example, the WTP estimates for salmon and steelhead are remarkably similar across regions and subspecies, with values ranging from $2.19 to $2.36 for an additional fish.

Figure 56. Marginal Willingness-to-pay, Based on a Meta-analysis of Recreational Fishery Valuation Studies



Source: Johnston *et al.* (2006).

Lost marine ecosystem services

Commercial exploitation of marine fisheries may also have adverse effects on non-targeted species (*e.g.* species at lower-trophic levels via food chains; disturbances caused by bottom-trawling on benthic ecosystems; genetic changes in fish populations due to selection), or may lead to a number of other non-fish related impacts (*e.g.* loss of non-fish marine biodiversity). It is exceedingly difficult to monetise the value of marine biodiversity, or the contribution of fishing practices to its loss.

Nevertheless, according to one UK study, the contribution of marine biodiversity to climate regulation has been valued at GBP 0.40-8.47 billion annually; the disturbance prevention and alleviation function has been valued at GBP 0.3 billion annually; and the value of marine biodiversity in terms of education and research opportunities has been evaluated at GBP 317 million annually (Beaumont *et al.*, 2006). Although other marine ecosystem services (resilience and resistance functions; bioremediation of waste; and provision of biologically-mediated habitat) may be equally important, no reliable valuation estimates for these services are currently available. The effect of fishing practices on these values is, of course, difficult to determine (Table 58).

Table 58. Value of Goods and Services Provided by Marine Biodiversity in the UK

Good/Service	Monetary value (2004 GBP per year)	Valuation method	Note
Resilience and resistance[1]	n.a.	n.a.	n.a.
Gas and climate regulation[3]	0.40-8.47 billion	Avoidance	Under-estimate
Bioremediation of waste (removal of pollutants)	n.a.	n.a.	n.a.
Biologically mediated habitat (habitat provision)	n.a.	n.a.	n.a.
Disturbance prevention and alleviation[4]	0.3 billion[5]	Avoidance	Under-estimate
Cognitive values (education, research)	317 million	Market	Over-estimate
Option use values (potential future uses)	n.a.	n.a.	n.a.

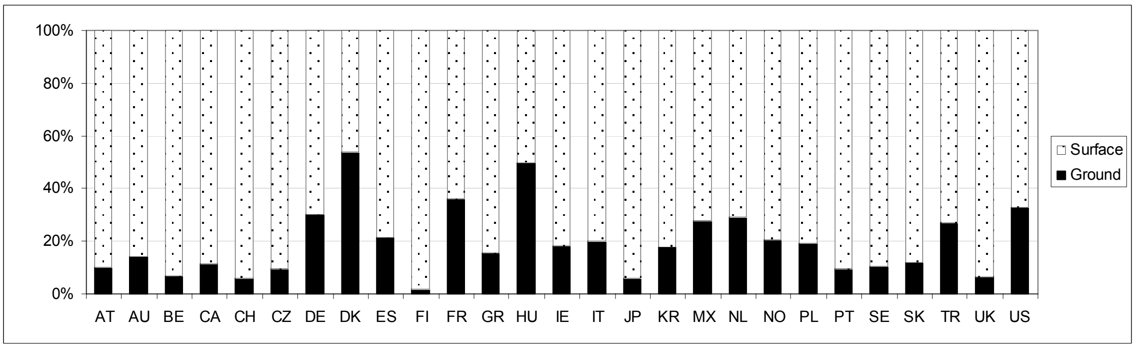
Notes: [1] The extent to which marine ecosystems can absorb recurrent natural and human perturbations. [2] Cost of treating UK waters once, not per year. [3] The balance and maintenance of chemical composition of the atmosphere and oceans by marine living organisms.[4] Dampening of environmental disturbances. [5] In addition to GBP 17-32 billion capital costs.
Source: Beaumont *et al.* (2006).

Groundwater Management

Introduction

Freshwater, of which groundwater is an important part, is a unique resource -- not only for its life-support (environmental) function, but also as a key input into almost any economic activity.[99] Groundwater accounts for over 97% of all freshwater available on earth (excluding glaciers and ice caps); the remaining 3% is composed mainly of surface water (lakes, rivers, wetlands) and soil moisture (EC, 2007a). The relative contribution of groundwater to a country's total freshwater endowment varies greatly across OECD countries – from less than 6% in Finland and Japan, about 27% in Mexico and Turkey, to more than 50% in Hungary and Denmark (Figure 57).

Figure 57. Endowment of Freshwater Resources in OECD Countries by Source, 2007



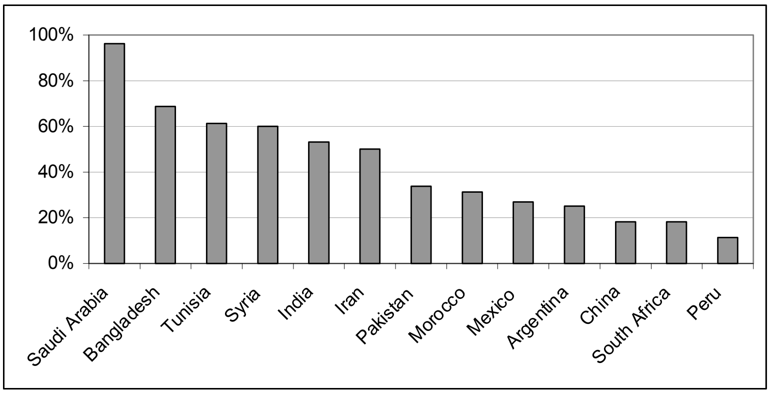
Note: The Figure shows countries' total endowment, not economically or technically extractable stock; includes internal water resources only.
Source: AQUASTAT.

Groundwater used for irrigation represents a major use of the resource. It has been estimated that irrigated areas supplied wholly or partly by groundwater have increased from approximately 30 million hectares in the 1950s, to close to 100 million hectares today (Shah, 2007). Global abstraction of groundwater for irrigation is estimated to be 900 cubic kilometers, having growing almost ten-fold in the last five decades (Shah, 2007). In the US, 81% of total consumptive use of groundwater is for irrigation (USDA, 2007). World-wide, about 40% of food grows in irrigated soils, some of which is supplied by groundwater extraction (Postel, 2001). Figure 58 provides an indication of the importance of groundwater for irrigation in selected countries.

Groundwater is also a major source of drinking water supplies. In the EU, 75% of the population depends on groundwater for their water supply (EC, 2007a). Many of the world's large cities also depend heavily on groundwater -- San Antonio, Texas and Tucson, Arizona are among the largest cities in the US that depend entirely on groundwater for their supply (Glennon, 2007; WSTB, 2002). Groundwater also provides about one-third of drinking water supplies in the Asia-Pacific region and Central and Southern America (Table 59).

99 See Barbier (2004) for recent work on the links between water availability and economic growth.

Figure 58. Percentage of irrigation water obtained from groundwater



Note*:* Definition of irrigation use and irrigated land varies between countries.
Source: UNEP (2003).

Table 59. Percentage of Drinking Water Supply Obtained From Groundwater

Region	% Groundwater	Population Served (Millions)
Asia-Pacific	32	1000-2000
Europe	75	200-500
Central and Southern America	29	150
USA	51	135
Australia	15	3
Africa	n.a.	n.a.
World	-	1500-2750

Source: UNEP (2003).

In OECD countries, the largest aquifers in terms of volume include the High Plains Aquifer (US) and the Great Artesian Basin (Australia). Some of the other economically important aquifer systems include the Alsatian and the Pannonian aquifers in Europe, the Denver Basin (Colorado), Central Valley (California) and Edwards aquifers (Texas) in the US, and the Valle de México aquifer system in Mexico. In many cases, the rate of aquifer recharge is extremely slow (Table 60). However, there is considerable uncertainty about the true volume of some aquifers. For instance, more recent estimates of Australia's Great Artesian Basin put the volume at 64,900 m^3 (Hillier and Foster, 2002).

Table 60. Some Large Aquifers of the World

Aquifer Name	Area (Million km^2)	Volume (Billion m^3)	Replenishment Time (Years)	Continent
Nubian Sandstone Aquifer System	2.0	75,000	75,000	Africa
North Sahara Aquifer System	0.78	60,000	70,000	Africa
High Plains Aquifer System	0.45	15,000	2,000	North America
Guarani Aquifer System	1.2	30,000	3,000	South America
North China Plain Aquifer Systems	0.14	5,000	300	Asia
Great Artesian Basin	1.7	20,000	20,000	Australia

Source: WWAP (2007) using data from Margat, 1990a, 1990b.

The relative importance of groundwater extraction and the rate of aquifer recharge determine the sustainability of groundwater management. For example, the High Plains (Ogallala) Aquifer in the US, which underlies parts of eight states (South Dakota, Wyoming, Nebraska, Colorado, Kansas, New Mexico, Texas, and Oklahoma), receives little recharge – especially in the southern portion. As a result of continued deep-well pumping, the resource has been declining steadily over the past three decades. A report by the

US Geological Survey (McGuire *et al*., 2000) estimated that water in storage in the High Plains (or Ogallala) aquifer in year 2000 was 6% less than the volume of water stored in the aquifer in the 1940s -- the time when significant groundwater pumping began. The greatest amounts of depletion during this time period were recorded in Texas (27% decline) and Kansas (16% decline). More recently, it has been estimated that the Texas portion of the aquifer is near depletion; water levels have declined 50-100 feet and well yields are down to 25% compared to the levels in 1980 (USDA, 2007). Groundwater depletion is a primary concern also in south central Arizona and the southern section of the Central Valley of California where groundwater overdraft has caused water table declines of 200 feet since aquifer development (USDA, 2007).

In Mexico, the number of over-exploited aquifers tripled between 1975 and 2004, from 32 to 104 (SEMARNAT, 2007). Table 61 gives an overview of the status of Mexico's aquifers. It indicates that, in some regions, as many as 20-30% of aquifers are being overexploited.

Table 61. Groundwater Exploitation in Mexico, 2004 (Number of Aquifers)

		TOTAL	Over-Exploited	%
I	Península de Baja California	87	7	8.0
II	Noroeste	63	18	28.6
III	Pacífico Norte	24	1	4.2
IV	Balsas	42	2	4.8
V	Pacífico Sur	38	0	0
VI	Río Bravo	96	16	16.7
VII	Cuencas Centrales del Norte	72	24	33.3
VIII	Lerma-Santiago-Pacífico	126	29	23.0
IX	Golfo Norte	42	3	7.1
X	Golfo Centro	22	0	0
XI	Frontera Sur	23	0	0
XII	Península de Yucatán	4	0	0
XIII	Valle de México y Sistema Cutzamala	14	4	28.6
	TOTAL	653	104	15.9

Source: SEMARNAT (2007), using data from CNA, Estadísticas del Agua en México 2005, México 2005.

Some of the 104 over-exploited aquifers exhibit exceptionally high extraction-to-recharge ratios, including those in the Central Valley, the Cutzamala system, Lerma-Santiago-Pacific, and the North Central Region (Table 62).

Table 62. The Most Unsustainably Exploited Aquifers in Mexico, 2004

Aquifer	Region		Index of Over-exploitation (Extraction/Recharge Ratio)
Texcoco	XIII	Valle de México y Sistema Cutzamala	9.57
Valle de la Cuevita	VIII	Lerma-Santiago-Pacífico	7.97
Vicente Suárez	VII	Cuencas Centrales del Norte	4.85
Monclova	VI	Río Bravo	3.60
Laguna Seca	VIII	Lerma-Santiago-Pacífico	3.10
Cuenca Alta del Rio Laja	VIII	Lerma-Santiago-Pacífico	2.95
Valle de Tulancingo	IX	Golfo Norte	2.85
Guadalupe de las Corrientes	VII	Cuencas Centrales del Norte	2.72
Coyotillo	II	Noroeste	2.71
Sonoyta-Puerto Peñasco	II	Noroeste	2.48
Cuautitlan-Pachuca	XIII	Valle de México y Sistema Cutzamala	2.38
Puerto Madero	VII	Cuencas Centrales del Norte	2.08
Salinas de Hidalgo	VII	Cuencas Centrales del Norte	2.07
Valle de Celaya	VIII	Lerma-Santiago-Pacífico	2.07
El Barril	VII	Cuencas Centrales del Norte	1.96

Source: SEMARNAT (2007), using data from CNA, Estadísticas del Agua en México 2005, México 2005.

In Europe, most countries for which data is available seem to have been exploiting their groundwater resources on a sustainable basis. Table 63 provides an indication of the rate of groundwater abstraction as a percentage of the available renewable recharge.

Table 63. Groundwater Abstraction in Selected European Countries (% of Available Resource*)

Country	1990	1996	2002
Belgium	-	74.6	-
Czech Rep.	62.4	46.1	40.3
Denmark	70.1	52.8	36.3
Greece	56.6	87.9	-
Cyprus[100] [101]	-	124.7[b]	44.8[c]
Latvia	-	36.2	23.3[d]
Hungary	-	13.0	12.9[c]
Malta	59.2	59.7	47.5[c]
Netherlands	54.7	60.7	47.6[d]
Austria	3.8	3.5	-
Portugal	76.6	157.3[b]	-
Slovakia	31.2	23.2	17.6
Finland	8.0	8.6[a]	-
Sweden	17.6	19.1	18.4
Romania	31.6	14.4	9.6
Turkey	30.7	43.6	48.8[d]
Iceland	-	2.6	2.5
Switzerland	1.9	1.7	1.7[d]

Notes*:* [a] 1995; [b] 1998; [c] 2000; [d] 2001; * Annual volume of abstracted groundwater is presented as a percentage of the resource available for abstraction over the long term (at least 20 years), calculated as groundwater recharge less the long-term annual average rate of flow required to achieve ecological quality objectives for associated surface water.

Source: EUROSTAT.

In a study of the Segura catchment area in Spain, Llamas (2003) emphasised the uncontrolled and unregulated nature of groundwater abstraction for irrigation. He pointed out that the productivity of areas irrigated from surface waters is much lower than that of areas irrigated from groundwater. As long as abstraction is uncontrolled, scarcity rents will be dissipated. Moreover, with recent technological advances (*e.g.* turbine pumps, more efficient drilling methods, and improved hydrogeological analyses), the rate of dissipation of these rents is accelerating.

In northern Africa, Libya's plans to extract groundwater from the Nubian Sandstone Aquifer, which underlies several North African countries, at the planned rate of 2.2 km^3 per year has been estimated to deplete the aquifer in 40-60 year time horizon (Postel, 1999). In China, the groundwater under the Huang-Huai-Hai plain has fallen by 50 metres in the last 35 years (World Bank, 2007). Parts of the aquifers in Hebei and Beijing are nearly dried up. In Punjab -- India's major agricultural production region -- groundwater levels have been dropping at 25-30 cm per year and the percentage of land where water table is below 10 metres has increased from 3 to 46 percent between 1973 and 1994. If the water table drops below 15 metres, the commonly-used tubewells will stop functioning (WRD, 2007).

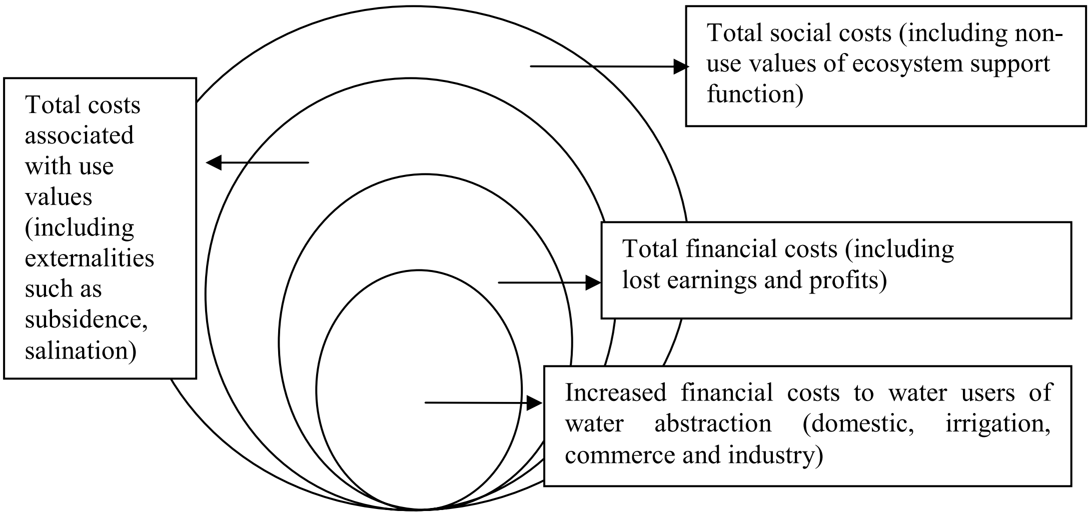
100 Footnote by Turkey: The information in this document with reference to « Cyprus » relates to the southern part of the Island. There is no single authority representing both Turkish and Greek Cypriot people on the Island. Turkey recognises the Turkish Republic of Northern Cyprus (TRNC). Until a lasting and equitable solution is found within the context of United Nations, Turkey shall preserve its position concerning the "Cyprus issue".

101 Footnote by all the European Union Member States of the OECD and the European Commission: The Republic of Cyprus is recognised by all members of the United Nations with the exception of Turkey. The information in this document relates to the area under the effective control of the Government of the Republic of Cyprus.

Aquifer pollution is fast becoming a concern. According to a report by the European Commission on groundwater pollution caused by nitrates from agricultural sources (EC, 2007b), 17% of EU monitoring stations (average values) had nitrate concentrations above the 50 mg/l limit in the period 2000-2003. Although large differences exist (depending on the depth of monitoring stations and the type of monitoring), the highest percentages reported by the study (20-60%) of groundwater sampling sites exceeding the limit value were reported in Belgium, the Netherlands, Luxembourg, Portugal, and Spain (EC, 2007b).[102] In Italy, excessive abstraction has caused deterioration of water quality in aquifers in coastal areas due to saltwater intrusion (Massarutto, 1999).

Figure 59 illustrates the different types of costs arising from unsustainable groundwater management. The innermost circle contains costs for user (domestic, agriculture, commerce and industry) directly associated with extraction (or use of an alternative source). The next circle contains the indirect impacts on regional economic activity, such as lost earnings of workers and foregone profits. The next circle includes economic non-pumping externalities which result in use costs (*i.e.* subsidence and salination). And finally, the outermost circle represents costs associated with damages to non-use values, such as the life-support function of water.

Figure 59. Social Costs of Inaction with Respect to Groundwater Management



Causes of inaction

There are several reasons why groundwater resources merit the attention of policy-makers. First, the total volume of groundwater stored in aquifers at any given time is finite. Thus, groundwater is a depletable natural resource, although in cases of significant aquifer recharge, use of the resource on a renewable basis is possible.[103] However, the total volume of groundwater stored in an aquifer and its recharge rate are

102 There were a total of approximately 20,000 groundwater montoring sample sites in the EU 15 in 2002-2003. The density of the network was on average 12.5 points per 1000 km^2. The network covers a large proportion of the groundwater sources in which nitrate pollution is likely to be a problem.

103 Groundwater has the characteristics of a *renewable* resource, if the rate of recharge of an aquifer is positive (*e.g.* many shallow alluvial or coastal aquifers with sufficient rainfall or surface runoff). These aquifers are extracted on a renewable basis as long as the rate of extraction is less than the recharge rate. However,

frequently *uncertain*. It can therefore be difficult to determine whether or not a particular use of the resource is sustainable.[104] In many cases, economic systems (cities, agriculture) have developed, without taking due account of the sustainability of the water resource upon which those systems depend. Indeed, the growth in groundwater use in many countries is essentially unplanned (Shah, 2007; and Llamas, 2003).

The main reason for this is the low degree of excludability associated with groundwater resources. Groundwater has the characteristics of a common-pool resource, so its exploitation will likely exceed its most economically efficient rate.[105] The inefficiency of unregulated open-access groundwater extraction is due to the private costs being less than the social costs of withdrawing groundwater. In other words, an individual user pays only the pumping costs and does not take into account the value of the water removed from the common pool. This can be exacerbated by other policy failures – *i.e.* subsidised energy which reduces pumping costs (Shah, 2007).

The external costs of pumping from a common aquifer include (Peterson *et al.*, 2003):

- *Stock cost* (the loss in water to future users for each unit pumped today);
- *Depletion cost* (the increase in irrigation costs as pumping today lowers the water table tomorrow);
- *Risk cost* (an aquifer provides an 'insurance' against variability in rainfall lowering water user's exposure to production risk).

However, the extent of external costs is not limited to increasing pumping costs due to falling water levels and well interference. Other important externalities exist, including (Brajer and Martin, 1989; and Llamas, 2003):

- *Land subsidence* (damage to surface and subsurface structures due to groundwater withdrawal);
- *Groundwater contamination* (reduction in groundwater quality, due to aquifer development, such as agricultural, industrial, or municipal runoff, which may cause aquifer contamination, or saltwater intrusion in coastal aquifers); and
- *Ecological impacts* (drying up of wetlands, disappearance of riparian vegetation because of decreased soil moisture, or alteration of natural hydraulic river regimes).

This is complicated by the fact that water is a unique natural resource and a strategic asset which cannot be substituted by other resources, below a certain point.[106] This distinguishing feature of water is typically not associated with other natural resources. It raises complex political and resource management issues. Since many urban agglomerations and agricultural systems are dependent upon continued extraction of

groundwater can be considered a *depletable* resource if an aquifer replenishes at a rate which is considered negligible on the human time scale. This is the case for many deep aquifers, aquifers isolated by impermeable layers, or those located in arid regions. The residence time of water stored in such aquifers spans over long periods of time (sometimes millennia). Extraction of such "fossil water" is sometimes referred to as "water mining".

104 See Llamas (2003) for an interesting discussion of the complexity of defining "overexploitation" of a groundwater resource.

105 Socially efficient extraction requires the price of groundwater be equal to the sum of the extraction cost and the user cost (scarcity rent) (see *e.g.* Hotelling, 1931).

106 This refers to "absolute scarcity" of water. In terms of "relative scarcity", substitution opportunities may exist (for example) by adopting more water-saving consumption and production patterns.

groundwater, it can be politically difficult to reduce extraction rates to a sustainable level. The loss of the very significant "sunk costs" involved is not usually politically practicable.

In the context of groundwater management, "inaction" can be best described as unsustainable resource management (*i.e.* where pricing of groundwater abstraction does not reflect its scarcity rent and the externalities). In practice, regulatory approaches to groundwater management often encourage, rather than constrain, groundwater abstraction (*e.g.* right of capture, "first-in-place" or "first-in-right" systems). Even fewer incorporate scarcity rents in water prices. However, in the absence of policies addressing groundwater pricing and access, the resource is likely to be extracted at a socially sub-optimal rate. Consequently, the costs of such policy inaction will result in a loss of social welfare.[107]

Costs of policy inaction with respect to unsustainable groundwater management

Why is inaction costly? Unregulated access to groundwater results in overuse of the resource. Initially, availability of groundwater in a region attracts migrants to settle and companies to invest. Over time, the growing number of migrants and the volume of economic activity leads to water shortages and eventually causes physical or economic depletion of the aquifer. Individual migrants or investors expect the government to ensure viability of the region. Paradoxically, this expectation becomes increasingly rational with the growing population size and the stock of investment committed in the area. This is because more population and higher value of the stock of investment increase the likelihood that the government will finance access to alternative sources in the face of aquifer depletion. Water that has taken millennia to accumulate may be used up in a very short period of time (World Bank, 2007).

More specifically, the cost of inaction is to expose a region to a potential loss of sunk costs, to adjustment costs, to increased economic vulnerability – *vis-à-vis* a key natural resource for which it is difficult to find a substitute.[108] Due to the low degree of substitutability of the resource, groundwater pricing which does not reflect the scarcity rent and externalities can lead to "resource lock-in", which requires continued government assistance and public funding. The "cost of inaction" therefore involves a vicious circle of population and investment increases, growing resource scarcity, and demands for government assistance (which becomes increasingly more difficult to deny with the growing scale of the problem).

Agricultural productivity and food security

In many countries, groundwater is an essential input of food production -- either as irrigation water or as input for the food-processing sector. Irrigated crops typically have significantly higher yields than dryland crops. In addition, dryland agriculture may be exceedingly risky (or infeasible) in some areas. Groundwater is particularly important in the areas of inland drainage which are not connected to the ocean including most of middle and central Asia and central Australia. Overall, irrigation (including irrigation using groundwater) has made a significant contribution to expansion of food production around the world and has been "a cornerstone of the green agricultural revolution" (WWAP, 2007).

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107 Under specific conditions, even with full property rights and due account taken of externalities, it may be economically optimal to "mine" groundwater.

108 For example, aquifer depletion may render the infrastructure of groundwater-dependent communities obsolete; aquifer depletion may increase economic vulnerability of groundwater-dependent communities -- by transforming a renewable resource into a finite resource; aquifer overdraft may expose groundwater-dependent communities to adjustment costs and the loss of "sunk costs" in the future. In addition, damages associated with aquifer overdraft and depletion may be irreversible, including land subsidence, groundwater contamination, and the loss of a non-renewable resource.

In many arid or semi-arid parts of the world, groundwater extraction has aided regional economic development and, if managed on a sustainable basis, can continue to do so into the future. However, unsustainable management of groundwater resources may create unwanted contingencies – dependence on a finite non-renewable resource. In many places, farming at the current extraction rates is temporary and eventually either the aquifer runs out of water or pumping from ever greater depths becomes prohibitively expensive. Water shortages will force farmers to look for alternative (more expensive) sources of water, switch to dryland cultivation, or abandon cultivation entirely and move elsewhere. All of these options carry a cost.

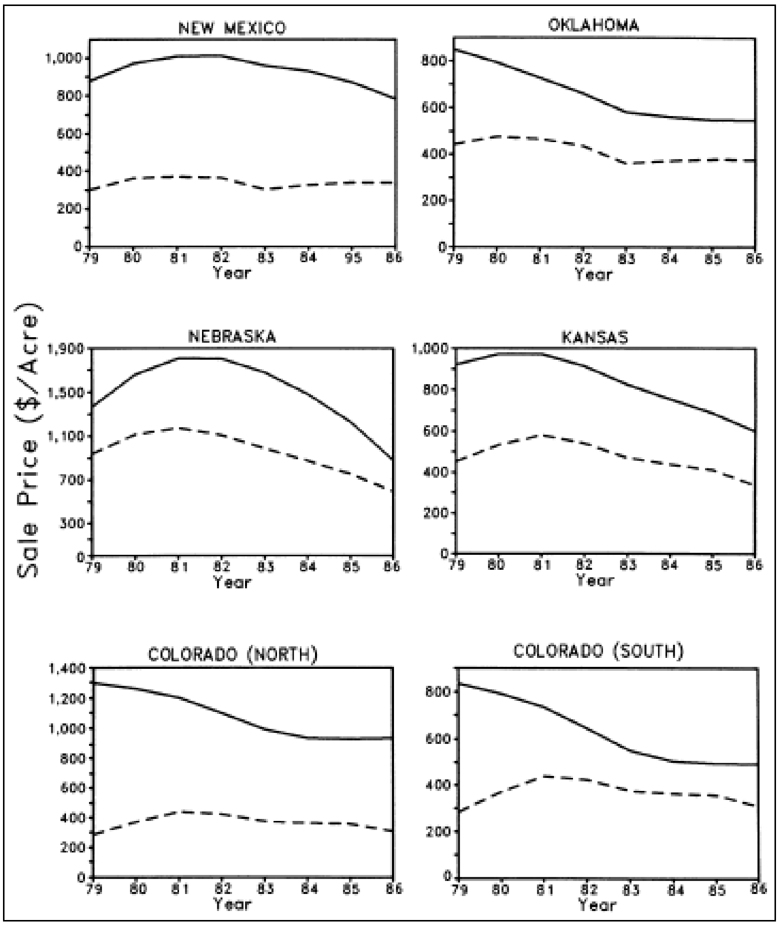
Declining groundwater supplies from the Ogallala are largely responsible for the loss of an estimated 1.435 million acres of irrigated cultivated cropland between 1982 and 1997 in the State of Texas (USDA 2007). In some areas, irrigated acreage is predicted to drop 50% by 2030 if current extraction rates continue (USDA, 2007).

Declining water levels mean higher pumping costs for farmers or, in the case of aquifer depletion, the need to switch to dryland agriculture. The return of irrigated farms to dryland production carries an economic cost for landowners, which is reflected in land values. Torell *et al.* (1990) used the price differential between irrigated and dryland farm sales observed in the marketplace to estimate the value of water in the Ogallala Aquifer. Controlling for differences in farm incomes, irrigation requirements, precipitation amounts, farm characteristics, and state-level variations in land prices over time, they calculated that the price differential between dryland and irrigated farms -- the market value of irrigation water -- had declined by different rates in different locations.

Figure 60 shows the estimated trends of irrigated (solid line) and dryland (dashed line) farmland values by state. The results indicate that the water value component of irrigated farmland price ranged from 30% to 60%, depending on location. Their results also suggest that the valuation of irrigation water in the Ogallala Aquifer (shown as the spread between the two curves) during the studied period had declined over time. The magnitude of the decline was estimated to be 30% in New Mexico, and as much as 60% in Nebraska and northern Colorado.

The relatively high value of water in New Mexico is related to the fact that New Mexico is an arid State, and dryland production involves much risk. While New Mexico farmers face limited cropping options without irrigation water, dryland cropping options in Nebraska, Oklahoma, and Kansas are more numerous (Torell *et al.*, 1990). Consequently, the estimated value of irrigation water in these States is less than in New Mexico (Table 64).

Figure 60. Declining Valuation of Irrigated (Relative to Dryland) Farmland



TIME TREND OF FARMLAND VALUES (—— IRRIGATED, — — — DRYLAND)

Source: Torell *et al.* (1990).

Table 64. Average Value of Irrigation Water as a Percentage of Total Irrigated Farmland Price (%)

	New Mexico	Oklahoma	Colorado (north)	Colorado (south)	Kansas	Nebraska	*Average*
1979	66	49	78	67	51	31	*57*
1980	63	41	70	54	45	33	*51*
1981	63	37	63	41	40	35	*47*
1982	64	34	61	36	40	39	*46*
1983	68	39	62	34	43	41	*48*
1984	65	34	61	30	41	41	*45*
1985	61	31	61	30	39	39	*44*
1986	57	32	67	39	43	32	*45*
Average	*64*	*38*	*66*	*44*	*43*	*37*	

Source: Torell *et al.* (1990).

In many countries, the increases in agricultural productivity which occurred throughout the 20th century have been possible due to the availability of "cheap" groundwater. The spread of irrigation was a key factor behind nearly tripling of global grain production since 1950 (Postel, 2001). Severe water scarcity thus represents the single biggest threat to future food production. Worldwide, about 8% of food crops grow on farms that use groundwater faster than the aquifers are replenished (Postel, 2001).

In China, the costs of groundwater depletion are significant, and much of this is borne by the agricultural sector. Taking into account salinisation and excessive rates of resource exhaustion, the costs have been estimated to be 50 billion RMB (World Bank, 2007). This is approximately 0.3% of GDP.

Urban settlement patterns and drinking water

Unsustainable use of groundwater in urban settlements may also have important economic consequences. Unlike irrigated agriculture, extraction of groundwater for municipal and industrial users typically involves significantly higher capital investment and population densities. Aquifer depletion and the consequent cuts in public water supply may cause disruptions to local economic activity, and the need for costly new water infrastructure.

For example, in the Mexico City Metropolitan Area, two-thirds of the water supply comes from the aquifer underlying the city and the rest from external sources, mainly from the Cutzamala River Basin. However, water supplied from the Cutzamala Basin incurs higher pumping costs due to its distance from the city (ca. 130 km) and lower altitude (1,000 metres below the city) (Saade, 2001). Many other Latin American cities depend to some degree on groundwater supplies (Table 65).

Table 65. Groundwater Dependence of Selected Cities in Latin America

	Dependence on groundwater
Managua, Nicaragua	100%
Havana, Cuba	100%
Mexico City, Mexico	80%
Guatemala City, Guatemala	70%
Lima, Peru	43%
La Plata, Argentina	40%
Quito, Ecuador	40%
São Paulo, Brazil	33%

Source: Anton (1993).

High population growth is projected to increase the dependence of many of these cities on groundwater supplies (Anton, 1993). An important feedback effect of urban expansion is that larger impermeable surfaces hinder aquifer recharge, and may thus further exacerbate existing water problems.

Contamination from agricultural non-point source pollution, pollution due to aquifer overdraft, and saline intrusion in coastal aquifers are major threats to maintaining groundwater quality. This can result in significant costs. For example, Koundouri and Pashardes (2002) attempted to value the effect groundwater salinity by comparing the price of land used in agricultural production and in tourism industry. They estimated that the farmers' average marginal WTP for avoiding coast proximity and related groundwater salinisation was US$ 55.21 per acre.

In many places, groundwater resources constitute a strategically important reserve of drinking water. For example, in the US, groundwater provides 35% of the public supply of drinking water and as much as 80% of drinking water in rural areas (Boyle *et al*., 1994). In the EU, groundwater provides about 70% of piped water supply (WWAP, 2007). Given potential impacts on households, the quality of groundwater resources is closely monitored. Concerns over the pollution of groundwater arise because of their potential adverse human health effects. Concerns over groundwater quality are most commonly associated with potential contamination from agricultural runoff as well as from municipal and industrial uses. However, groundwater quality in coastal aquifers may also be degraded as a result of saltwater intrusion due to excessive withdrawals. Table 66 gives an overview of studies conducted to value human health impacts of groundwater contamination.

Table 66. Valuation of Groundwater Protection for Drinking Water Supplies

Study	Mean WTP per household per year	Stressor	Method	Location
Hasler *et al.* (2005)	EUR 255*	Resource extraction; Toxins	CE	Denmark
Martin & Marceau (2001)	CAD 48, CAD 78	Scarcity; overuse; pollution	CV, AB	Canada
Poe & Bishop (1999)	USD 412 [1]	Nitrates; Agriculture	CV	US
Stenger & Willinger (1998)	94-110 €(1993)* for users; 52-90 €(1995)* for non-users	Chemicals/toxins; Agriculture, landfills, transportation, *etc.*	CV	France
Giraldez & Fox (1995)	CAD 693-6289	Nitrates; Agriculture	AB	Canada
Press (1995)	EUR 207-516*	Agricultural chemicals	CV	Italy
Bergstrom & Dorfman (1994)	USD 296-613 if low probability USD 1162-2360 if high probability of pollution	Bioaccumulative and toxic substances; Agricultural pesticides and fertilizers	CV	US
Jordan & Elnagheeb (1994)	USD 10-49 per month	Nitrates; Agriculture	CV	US
Powell *et al.* (1994)	USD 62	Bioaccumulative and toxic substances; Infrastructure development	CV	US
Poe & Bishop (1992)	USD 257-415	Nitrates; Agriculture	CV	US
Shultz & Lindsay (1990)	USD 129-215	Chemicals/toxins; Stationary sources	CV	US
Powell & Allee (1990)	USD 42-81	Chemicals/toxins; Agriculture, landfills, sewage, *etc.*	CV	US

Notes: CV = Contingent valuation method; AB = Averting behaviour method; CE = Choice experiment
* Approximate value in Euro.
[1] Maximum WTP for an incremental benefit of 25% reduction in nitrate exposure.
The estimated values of WTP are not directly comparable due to differences in methodology, scenarios, and sample characteristics (*e.g.* household income and location) across the different studies.

Several studies have estimated option prices associated with avoided contamination of groundwater supplies. For example, Sun *et al.* (1992) analysed the willingness-to-pay of groundwater users for a groundwater contamination abatement programme in the face of uncertainty about potential contamination by agricultural chemicals. The mean option price for groundwater quality protection has been estimated at $641 per year. In another study, Edwards (1988) estimated the aggregate benefits to prevent uncertain, future nitrate contamination of groundwater supply from sewage and agriculture at $5-25 million per 1000 households, representing the present value of a stream of benefits over a 30-year time horizon using a 4% discount rate.

Land subsidence and loss of ecosystem functions

Aquifer overdraft may cause land subsidence causing damages to surface and subsurface structures. In addition, land compaction may permanently damage an aquifer because full recharge of a depleted aquifer will not be possible. The potential impacts on aquifer storage and conductivity may be irreversible, leading to a loss of the resource.

For example, depletion of an aquifer under Mexico City has caused land subsidence in the city centre, estimated at 7.5 metres on average over the course of the last century, with some areas sinking by as much as 2 metres in the last decade (Saade, 2001). In the US, land subsidence has affected approximately 8,500 square miles of land. The maximum subsidence was observed in San Joaquin Valley in California where the surface fell approximately 29 feet between 1926 and 1972 (Brajer and Martin, 1989). Land subsidence caused by aquifer depletion has reached 18 feet in some areas near Phoenix, Arizona (USDA, 2007). Serious problems due to land subsidence associated with groundwater extraction have occurred also elsewhere in the world, including Bangkok, Shanghai and Venice (Anton, 1993).

Groundwater is a key factor of river flow and riparian health, providing the base flow for surface water systems and acting as a buffer through dry periods. In many rivers, more than 50% of the annual flow is derived from groundwater. In low-flow periods in summer, more than 90% of the flow in some rivers may come from groundwater (EC, 2007a). Aquifer discharge thus contributes to maintenance of terrestrial, riparian, wetland and stygian ecosystems that may be entirely, or partially, dependent on groundwater (see *e.g.* Murray *et al.*, 2003). Since riparian and wetland ecosystem are typically characterised by a high degree of biodiversity, groundwater abstraction and deterioration of groundwater quality may cause biodiversity losses in such aquifer-dependent ecosystems.

Few studies have attempted to provide monetary estimates of the *ecological value* of groundwater. In Denmark, a choice experiment was conducted to estimate the willingness-to-pay of households to protect groundwater quality in order to provide "very good conditions for plant and animal life", estimated at DKK 1204 (approx. EUR 162) per household and year (Hasler *et al.*, 2005).

Summary

The costs of inaction with respect to natural resource management arise as a consequence of a rate of exploitation of the resource which does not optimise returns. This arises because, in many cases, it can be difficult to exclude potential users from exploitation of the resource. The creation of effective property rights over the resource and/or effective regulation of access and exploitation is likely to result in a more efficient management of the resource.

Unfortunately, many of the world's fisheries and aquifers are being exploited at unsustainable rates, even though over-exploitation has been identified for many years as a major problem in both cases. In the case of fisheries management, there are already examples of fish stocks which have been driven to commercial extinction. In the case of groundwater, there are many examples of "mining" of aquifers, with the development of urban agglomerations and agricultural systems becoming more dependent upon a depleting resource.

The costs of unsustainable fisheries resource management can be considerable. In addition to the direct costs associated with the loss of the commercial value of the stock, there are a number of important secondary impacts, including loss of employment and increased government expenditures to compensate for impacts on local communities. The costs in terms of non-market impacts (such as recreational impacts and loss of marine biodiversity) are difficult to value, but are also likely to be considerable.

Reduction in fishing pressures, before stocks are driven to very low levels, would allow the stocks to recover, and thus enable higher sustainable yields in the future. However, due to the nature of the resource, fisheries management takes place against a backdrop of imperfect information and imperfect control. The size of the stock, its growth rates, and its relationship with other stocks are not known with precision. And even if they were known with precision, regulation of the sector is imperfect, particularly in some areas (*i.e.* high-seas fisheries). In the face of imperfect information and control, precaution should be exercised since if thresholds are breached a stock can be fished into commercial extinction, with the permanent loss of all of the benefits set out above.

The costs of excessively rapid exploitation of groundwater resources can also be considerable. Freshwater availability is one of the most serious long-term problems confronting many OECD as well as non-OECD countries. In many parts of the world, water shortages may be the "limiting factor" of economic development. While groundwater extraction can help to relax this constraint, it is important to incorporate scarcity rents in water pricing and thus match the rate of groundwater extraction with the available resource base.

If these scarcity rents (and associated externalities) are not reflected in the price of water to users inefficient and unsustainable patterns of agricultural and urban development can come into being. Ultimately, given the low substitution possibilities for water and the high fixed costs associated with such patterns of development, this can result in a form of "lock in", in which it becomes increasingly difficult to adopt more sustainable management practices, even as the costs of inaction rise.

CHAPTER 6. SUMMARY AND CONCLUSIONS

General conclusions

Estimating the costs of inaction on key environmental challenges is important because it allows policy-makers to better understand the nature and scope of these challenges, making it easier to decide when (and how) to intervene with policy. This is particularly important in the environmental field, because so many of the impacts of inaction in this field are not reflected in markets.

This report summarises available evidence on the costs of inaction in four key areas of environmental policy:

- Air and water pollution effects on human health;
- Climate change;
- Environment-related industrial hazards and natural disasters; and
- Natural resource management.

Despite the measurement difficulties, the existing literature suggests very strongly that the costs of policy inaction in selected environmental areas can be considerable -- in some cases, representing a significant "drag" on OECD economies. A few examples include:

- Stern's (2007) "best estimate" of the discounted value of the costs of not introducing policies to mitigate climate change is 14.4% in terms of per capita consumption equivalents. Others (*e.g.* Nordhaus, 2007) estimate much lower costs. However, there is broad agreement that climate change will have significant economic consequences.

- Muller and Mendelsohn (2007) have estimated that the total damages associated with emissions of air pollution from 10,000 major sources in the US are between \$71 billion and \$277 billion (0.7% to 2.8% of GDP).

- In the case of China, these costs are expected to be even higher. According to World Bank (2007), the health impacts associated with air pollution in that country are about 3.8% of GDP, with much of these impacts occurring in urban areas. Water pollution costs in China may also represent between 0.3% and 1.9% of rural GDP (depending on the "value of a statistical life" that is applied).

- The costs associated with oil spills can be significant. Carson *et al.* (2003) estimated the social cost of the *Exxon Valdez* spill at \$2.8 billion. In Europe, the costs to Galicia of the *Prestige* spill have been estimated at approximately EUR 567 million (more than 1.5% of annual GDP).

- Bjørndal and Brasao (2005) have estimated that net present value associated with retaining the existing fishery management regime (*i.e.* total allowable catch and gear selection) for East Atlantic Bluefin tuna is one-third what would be achieved from an optimal regime, resulting in a loss of \$US 1-3 billion.

- The World Bank (2006c) has estimated that, for the poorest countries, the cost of natural disasters represents more than 13% of GDP. Although only some of this amount is attributable to environmental factors which can be influenced directly by public policy, the proportion is likely to be increasing over time.

- It has been estimated that salinisation of groundwater affects agricultural productivity on 22 million ha of land, particularly in China, India, the Commonwealth of Independent States, the US, and Pakistan. The farmers affected by this problem lose up to $11 billion per year as a result (UNEP/DFID/DGDC/BGS, 2003).

- The costs of not meeting existing international commitments for water and sanitation (*e.g.* halving the proportion of the population who do not have access to improved water and sanitation) have been estimated at $128.9 billion per year (Hutton and Haller, 2004).

Defining and measuring the cost of inaction is complex – partly because of the environmental and economic uncertainties involved; but partly because of difficulties in establishing both the baseline and the boundaries for these estimates. For example, some of the costs of inaction will be incurred locally (and immediately), while others will fall on citizens in other countries (and perhaps in the distant future). Similarly some costs will be reflected in very tangible form (e.g. expenditures on health services), while others will be more intangible (e.g. “increased pain and suffering”)

Other elements of the costs of inaction are less apparent (and more difficult to quantify) – such as the costs associated with the loss of marine and terrestrial biodiversity. Still other elements of the total cost of inaction include intangible and subjective costs, such as “pain and suffering” from ill-health. Other components of the costs of inaction may be reflected in existing markets, even though they are not readily perceived as costs of environmental policy inaction *per se*. Examples include the effects of contaminated sites on adjacent property prices, or the effects of air pollution on agricultural yields. “Previous inaction” may also have left an important negative legacy in some problem areas (*e.g.* contaminated sites, accumulated stock of GHGs, unregulated groundwater extraction). All of these components are important for policy discussions that focus on the costs of inaction.

It is clear that OECD countries have made significant strides in addressing many of the environmental concerns discussed in this report. The term “inaction” must therefore be interpreted in this context. Even if the full costs of inaction are deemed to be significant, identifying those areas in which new environmental policies should be undertaken would still require a careful balancing of the marginal costs of inaction with the marginal costs of further reducing the associated impacts beyond those measures already in place. Although an assessment of some of the elements of one side of this equation is instructive, this important additional step would also need to be taken, before arriving at policy decisions.

Given the uncertainties involved, and the fundamentally tendentious nature of the problem of estimating the costs of inaction, it would therefore be foolhardy to attempt to develop an aggregate estimate of the cost of environmental policy inaction.

Nor has any effort been made here to summarise available evidence associated with the costs of setting more ambitious environmental goals, or to consider overall policy *priorities* – although a review of the evidence concerning the magnitude, incidence, and form of the costs of inaction on environmental problems is clearly *one* important input to those policy discussions.

OECD governments have, for many years, developed policies to address these environmental challenges however, much work remains to be done. In particular, work should be intensified to reduce some of the uncertainties involved in defining and measuring the marginal costs of *inaction*, so that eventual comparisons with the marginal costs of *action* can be as robust as possible.

Key methodological issues

There are several key methodological issues that arise when seeking to estimate the costs of inaction in the environmental policy domain. The most important of these problems are:

Uncertainty and Imperfect Information

There are some types of environmental pressures for which the costs of impacts are very uncertain. For instance, fisheries managers only have imperfect information concerning the status of fish stocks, the effects that different levels of fishing effort will have on the stocks, and the future stream of benefits associated with future yields of commercial fish stocks. Similarly, there is uncertainty about the effect that a given level of GHG emissions will have on GMT, the effect of GMT on tropical storms, the damages (health and material) that will arise from such storms, and the valuation of these damages. The review of the aggregate estimates of the costs of not mitigating GHGs reveals that the results of different (credible) studies can differ by an order of magnitude. In some cases, there is even ambiguity about the sign of the impacts.

There is, therefore, considerable uncertainty associated with all stages in the "costing" of the impacts of the environmental and resource degradation. This raises different concerns which are of direct relevance to valuation. First, it is important to undertake new research to reduce the level of uncertainty. Second, it is important to reflect this uncertainty in the valuation studies that are undertaken and in the way the results of these studies are communicated. In the presence of significant uncertainty, it is important to assess how much this uncertainty affects the range of possible "costs".

It may not even be possible to assign probability distributions to different environmental outcomes. There are some types of potential impacts where "we do not even know what we do not know" (Cole, 2007). In such cases; it may be inappropriate to use standard valuation methods which are based on certainty-equivalence (*i.e.* probability-weighted outcomes). Some catastrophic events potentially arising from climate change fall into this category.

Irreversibility and thresholds

In addition to uncertainty, there are several areas in which environmental pressures have potentially "irreversible" consequences. Examples include:

- Oil spills and loss of some local ecosystem functions and biodiversity;
- Bio-accumulative health impacts associated with water pollution;
- Extraction of groundwater sources, leading to "collapse" of the aquifer;
- Overfishing and commercial extinction of a fish stock; and
- Climate change and deglaciation of ice sheets.

In the presence of these irreversibilities, the costs of policy inaction must include the cost of losing the potential benefits of exploiting the resource at any time in the foreseeable future (*i.e.* the "option" value). Option values can dominate other elements in the estimated costs of inaction, if the potential irreversibilities are catastrophic. Even the option value associated with the (non-catastrophic) permanent loss of a fish stock can be significant, outweighing other costs of inaction (Leon *et al.*, 2003).

The policy implications of irreversibility are closely related to those arising out of uncertainty. Indeed, there are no specific policy implications arising out of irreversibilities in the absence of uncertainty. In effect, irreversibilities magnify the importance of uncertainty. Pindyck (2007) illustrated this point with reference to the difference between *flow* (*e.g.* particulate matter) and *stock* (*e.g.* carbon dioxide) pollutants. Since past contributions of stock pollutants to present and future concentrations cannot be "undone", any uncertainty about their potential impacts will have greater potential implications on estimated costs. There is a "ratchet effect", which makes the costs of "bad news" greater than the benefits of "good news".

Long-run and discounting

Environmental concerns have brought important issues related to the treatment of future generations into sharp relief. Optimal natural resource management in the area of fisheries, forestry, and groundwater extraction requires significant foresight, with implications extending over decades. Even in the absence of catastrophic and irreversible impacts, assessments of climate change impacts require policy planning horizons which may extend centuries into the future.

With impacts extending over long periods, it is necessary to express the costs of inaction borne far in the future in a manner which is commensurable with costs borne today.[109] A given impact borne today should not be valued the same as one borne in the future, both because of the opportunity cost of capital and because of people's time preference. Estimates of the costs of inaction can vary significantly, depending on the discount rate applied. The case of climate change is illustrative (Stern, 2007a; and Nordhaus, 2007).

This point is also relevant for issues such as natural resource management and latent health impacts. As noted in Hepburn (2007), the application of a 6% discount rate (rather than a 3.5% rate) increases the estimated costs of inaction with respect to PM almost three-fold. Due to its slow growth, a Scottish Oak plantation does not generate positive benefits with a 3.5% discount rate – but it does do so if time-varying (decreasing) discount rates are assumed over the life of the project.

Therefore, the choice of discount rate matters; and this choice is usually controversial. For impacts which are incurred in the very distant future, there is some question as to whether people's rate of time preference (*i.e.* revealed "impatience") should be considered a legitimate reason to discount costs. In the face of uncertainty concerning future interest rates and the future path of the economy, some have argued for the use of a discount rate which declines through time (Weitzman, 2002).[110] Depending on the degree of uncertainty involved, this value may converge on a low discount rate.

Whatever the choice of discount rate, the appropriate value will *not* be zero. With zero discounting, societies will devote an overwhelming proportion of today's resources toward avoiding any impact which incurred high up-front costs, but which provided a stream of benefits which continued indefinitely. With growing economies, people will be wealthier in the future than they are at present, and this would imply a significant transfer of wealth from relatively poorer (present) generations to relatively wealthier (future) ones.

Substitutability and sustainability

The estimated costs of inaction of environmental degradation depend in large part on the extent to which the environmental resources affected can be substituted. In a global sense, environmental resources are, of

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[109] As well as in a manner which is commensurable with the costs of addressing the environmental problem over long time horizons.

[110] Again, this approach has only been adopted by a small minority of OECD country governments in project evaluation.

course, not substitutable. There is no limit that can be reasonably placed on value of global ecosystem functions, so there is no limit to the estimated costs of any inaction which leads to their destruction.

However, there are significant differences in the extent to which individual resources can be substituted, and the extent to which they are substitutable at a local level. Aquaculture fish production *is* being successfully substituted for (heavily exploited) marine capture fisheries, for example. However, the growth of aquaculture is also itself quite dependent on the continued health of marine fish stocks. Planting of drought-resistant crops can also compensate for loss of water resources arising out of groundwater depletion or climate change. However, a minimum level of water is a precondition for cultivation.

Up to a point, economic sustainability is compatible with the substitution of environmental resources for other inputs. However, for many types of resources there are limits below which further substitution results in devastating economic loss. The less easily substitutable is the resource the greater is the "cost of inaction" associated with its exploitation, and the less sustainable is a path of development which involves its degradation.

Equity and distribution

The areas chosen for review in this report clearly illustrate that environmental impacts can affect different populations very differently. For instance:

- There is good evidence that residents of poorer countries will be particularly affected by climate change and will be less able to adapt to its impacts;

- It is frequently poorer households which are particularly affected by local environmental "bads", such as local air pollutants and contaminated sites; and

- Local communities are frequently most affected by the depletion of resource stocks (fisheries, groundwater) upon which they are dependent.

The distribution of impacts (and not just their magnitude) is relevant to policymakers for a number of reasons. First, for many of environmental issues, the "costs" are sufficiently important that there will be very significant impacts upon relative wealth within and across countries. Second, for those impacts which extend across countries or generations, there may be no means by which the "winners" can compensate the "losers". With climate change, for example, the transfer of wealth arising out of the adverse effects of emissions from some countries (those with significant emissions) on other countries (those heavily affected) is likely to be much greater than existing flows of Official Development Assistance.

The "weighting" of impacts has been proposed by research as a means to take distributional impacts into account when assessing the costs of inaction, the idea being that by attributing a greater weight to impacts which affect the poor than the rich, society's aversion to inequality can be reflected directly in the estimated costs of inaction. However, it is important to recognize that this approach can have a significant impact on the aggregate estimates of the estimated costs of inaction.[111]

[111] The UK *Green Book* (2003) is one of the few government documents which provide policy evaluation guidelines on the issue of equity weighting. It notes the practical complications associated with applying such weights, but concludes that appraisers should, "where deemed appropriate, attempt to adjust explicitly for distributional implications". Again, this is not a view shared by all OECD country governments.

Endogeneity and adaptation

Valuing the costs of environmental policy inaction depend on an understanding of how households, firms, farmers, etc. are likely to respond in the face of changing environmental conditions. This "adaptation" can take many forms:

- With changing temperatures and precipitation due to climate change, farmers may change input choice, crop selection and tilling practices;

- With rising sea levels and more frequent extreme weather events, there are likely to be new investments made in protective infrastructure and changing development patterns;

- With local air pollutants or contaminated sites, household choices of residential location will be affected; and

- With groundwater depletion, alternative sources of water (and means of livelihood) will be explored.

Assuming that households, firms and farmers are "myopic" is, of course, unrealistic, and will likely result in a significant *over*estimate of the costs of inaction. Work in the area of agriculture, for example, has indicated that this overestimation can be significant -- often more than 50% of the estimated costs (Fankhauser, 2006).

On the other hand, assuming that households, firms and farmers have "perfect foresight", and are able to adjust costlessly to changing environmental conditions, will result in *under*estimates of the true costs of inaction. For one thing, information is unlikely to be perfect -- it can be difficult to distinguish between normal random fluctuations in environmental conditions and long-term trends. There are also significant costs likely to be associated with investing significant resources to adapt to changes which prove to be transitory. Analogously, there are costs associated with mistaking underlying change for a temporary phenomenon.

Perhaps more importantly, even if there are no information problems, there may be market barriers and failures which constrain efficient adaptation. As noted earlier, these problems are likely to affect residents of developing countries relatively more acutely than OECD countries. With limited sources of savings and imperfect capital and product markets, adaptation in the former countries will be more limited than would be optimal. Providing support for adaptation in the face of such information and market failures would therefore likely reduce the costs of inaction.

Different types of "cost" arising out of inaction

Evaluating the costs of inaction involves estimating the total economic value (use and non-use) of a given deterioration in environmental quality. However, the precise form in which the costs of inaction associated with environmental degradation are reflected in the economy (or directly in people's welfare) varies widely. Some of the most important impacts are reflected as *direct financial costs for productive sectors of the economy*. Examples include lost yields of commercial fisheries due to unsustainable stock management, increased expenditures on water treatment infrastructure due to water pollution, and lost land resources due to sea-level rise from climate change.

These costs can be considerable. For instance, groundwater depletion (or pollution) can have significant impacts on agricultural yields -- due to reduced irrigation possibilities. In some cases, groundwater depletion may even render existing agricultural land unviable. It has been estimated that between 1982 and

1997, 1.435 million acres of irrigated cropland in Texas were brought out of cultivation, due to ground water depletion (USDA, 2007).

In the case of natural disasters, some of the most visible costs of inaction relate to the need to reconstruct damaged physical infrastructure. Focusing only on property damages there have been 10 hurricanes which have caused damages in excess of $US 10 billion, with five of these occurring in the last decade (Blake *et al.*, 2007). While the relationship between anthropogenic factors, such as human-induced climate change, and the frequency and intensity of extreme weather events cannot de determined with precision, this gives at least some indication of the potential costs involved.

Even the "first-order" restoration and clean-up costs associated with oil spills can be significant. In the case of the *Erika*, these direct costs were estimated to be EUR 100 million (Bonnieux and Rainelli, 2003); for the *Prestige*, they were valued at over EUR 500 million (Loureiro *et al*., 2006 and Garza-Gil *et al.*, 2006). For the *Exxon Valdez*, clean-up costs alone were over $2 billion (Carson *et al.*, 1992). This ignores all the other impacts of oil spills, such as effects on ecosystems, on the fisheries sector, and on tourism, which are often expected to be significant.

There are also significant *indirect market impacts*. While these costs also affect productive sectors of the economy, they are often more difficult to quantify with precision. For instance, the impacts of environmental factors on factor productivity can be considerable. With respect to air pollution, several studies report on the negative effects of O_3 pollution on yields. In Europe, for example, it has been estimated that the costs of not having introduced the Gothenburg Protocol in terms of agricultural output alone would have been EUR 462 million/year (Holland *et al.*, 2002).

Analogously, the impacts of air and water pollution on labour productivity can also be considerable. For example, Samakovlis *et al.* (2004) estimated that an increase of 1 $\mu g/m^3$ in NO_2 emissions in Sweden resulted in a 3.2% increase in respiratory-related restricted activity days – approximately 685,637 additional restricted activity days. In a Norwegian study, Hansen and Selte (2000) found that the effect of reducing PM_{10} concentrations in Oslo from 24.5 $\mu g/m^3$ to 12.3 $\mu g/m^3$ would reduce the sick leave ratio by 7%. In developing countries, the time "lost" in an effort to secure clean drinking water is also very considerable -- with associated impacts on schooling and employment.

Just as the productivity of certain sectors and factor inputs can be affected by environmental degradation, the "quality" of marketed goods and assets may be significantly affected. The case of real estate markets is indicative. For instance, in a study of the Chesapeake Bay, Poor *et al.* (2007) found that a one mg/litre increase (approximately 8%) in total suspended solids resulted in a fall in coastal property prices of $1,086 (approximately 0.5%). For dissolved inorganic nitrogen, a one mg/litre change (300%) resulted in a $17,642 fall (approximately 9%). Gibbs *et al.* (2002) found that a one-metre decrease in underwater visibility in New England led to a decrease in property value of 6%.

While these impacts are mainly local and/or sector-specific, some environmental impacts may be of such importance as to affect *the macroeconomy more generally*. This is most likely to be the case with climate change. Climate change may impact on aggregate levels of investment and savings, which affect the entire economy. In one of the few studies to look at the effects of climate change on important macroeconomic fundamentals, Fankhauser and Tol (2005) carried out simulations which took into account the prospect of future damages on capital accumulation and savings rates. They found that these "indirect" costs can even exceed the "direct" costs of climate change -- with the difference becoming greater over time.

In the face of rigidities in capital and labour markets, the costs are likely to be greater still, particularly if the change in environmental quality is sudden. Using a model which allows for market rigidities in the adjustment to an extreme weather event "shock", Hallegatte *et al.* (2006) found that the overall impacts are much greater than if a smooth adjustment is assumed (as is the case in many models). Ultimately, with sufficient extreme weather event activity, an economy may find itself in "perpetual reconstruction", with the economic impacts again being amplified over time.

More subjective and *intangible* impacts may also be important, even if difficult to estimate. The costs of "pain and suffering" associated with environmentally-induced ill-health are illustrative. In a study of acute cardio-respiratory morbidity cases in Canada, Stieb *et al.* (2002) estimated that, for some impacts (*e.g.* emergency department visits, asthma symptom days), "pain and suffering" represented 40% or more of the total health costs of particulate matter. In a French study, Rabl (2004) found that, for other types of impacts attributable in part to pollution levels (*e.g.* cancer), the proportion of costs represented by "pain and suffering" may even exceed 90%.

The loss of *non-use values* associated with environmental degradation can also be difficult to estimate. For instance, while this report has not directly reviewed the costs of inaction associated with biodiversity, estimates of the "existence" values which people attach to different species indicate that non-use values can be very significant (Stevens *et al.*, 2002).

While the actual estimation of such impacts is controversial, focussing on the costs of inaction without taking into account issues, such as the existence value of biodiversity or the "pain and suffering" that results from ill-health, can result in a gross underestimate of the costs of inaction. However, in some cases, the assessment of the more tangible market impacts alone may be sufficient to warrant additional policy interventions (*i.e.* above and beyond those policies that are already in place). Since these "more direct" costs are often easier to estimate with confidence, this is important to bear in mind.

Incidence

Ultimately, all of the costs of inaction fall on households, whether as residents, consumers, or taxpayers. However, the initial point of incidence of these costs is politically important, and this is likely to depend upon specific institutional factors which exist at the local or national levels. For instance, the share of costs borne by the private and public sectors will vary by type of impact and by country. Health costs associated with air or water pollution provide an example. The direct incidence of financial costs arising out of respiratory problems include at least: personal impacts in terms of "pain and suffering"; private preventive expenditures and medicine costs; public health service costs; and lost productivity at work. The extent to which costs are reflected in each of these categories is likely to vary widely across countries.

In a study of respiratory problems from air pollution in the US, Chestnut *et al.* (2005) distinguished between costs which are borne directly by the victim and those borne by third parties (caregivers, taxpayers, etc). It is interesting to note that the proportion of such financial and opportunity costs (ignoring "pain and suffering") borne directly by the individual sufferer in that study was less than 75% of total costs.

However, these percentages will depend on prevailing markets and policy factors. For instance, in Norway, full compensation is granted for sick leave for as many as twelve days per year, even without a medical attestation (Hansen and Selte, 2000). Perhaps more significantly, the balance between costs of health services will vary widely across countries, with costs being borne by the patient (out-of-pocket or insurance premiums) or the taxpayer to very different degrees. These differences clearly affect the "first-order incidence" of financial costs of inaction.

As noted earlier, the existence of contaminated sites represent a significant "legacy" of environmental costs of inaction in the past in many OECD countries. It has been estimated that annual remediation expenditures in Europe can be as much as 0.3% of GDP; and the undiscounted cumulative costs of remediation of contaminated sites represents approximately 2%-4% of a single year's GDP (EEA/CSI, 2005). However, there is wide variation in the incidence of these costs. In some countries (*e.g.* Czech Republic and Spain), the costs are borne entirely by the public sector; in others (*e.g.* France and Italy), more than 95% are borne by the public sector.

The case of coastal floods arising (in part) from climate change also highlights the importance of cost incidence. The extent to which households are compensated for losses depends in part upon the "insurance density", and this varies widely across and within countries. For example, the data suggest that the ratio of insured losses to overall losses has been about 38% in the US, versus about 27% in Europe during the period 1980-2005 (OECD, 2006a). However, these figures vary by incident. While "insurance density" in the US is thought to be about 25-50% (OECD, 2006a), in the case of Hurricane Andrew, the relevant figure was approximately 65%. For Katrina, it was 27-33% (OECD, 2006a). The extent of insurance coverage can affect rate at which reconstruction is undertaken, and thus, the adjustment costs.

Those who exploit a resource are often those who bear the highest cost from unsustainable management regimes. However, others may also bear some of the costs, including taxpayers. In response to the collapse of the cod stock in Canada, for example, substantial public funds were spent on income support (including fishers' unemployment benefits) and government assistance programmes (expenditures towards restructuring, sectoral adjustment, and regional economic development). An estimated CAD 3.5 billion was spent on these programmes (OECD, 2006c).

Differences in the direct incidence of costs can have important policy implications. On the one hand, it can affect the political attention received with respect to a specific concern. Arguably, costs of inaction that fall on public sector expenditures (reconstruction, health services, preventive measures, income compensation, etc…) will receive more political attention than those which have "first-order" effects on the private sector. However, even if the costs of inaction are borne by private firms and households, there can be significant variation in the political attention they receive. Costs which are borne "diffusely" may have less "resonance" in subsequent policy discussions.

On the other hand, the point of incidence of the costs of environmental policy inaction has direct implications for incentives to avoid future negative environmental legacies. Inaction is a reflection of the non-internalisation of environmental externalities. It is important that price and regulatory signals which reflect the costs of inaction be transmitted to those in a position to reduce such impacts, since *ex ante* prevention is often much less costly than *ex post* remediation or adaptation. In many cases (climate change, high-seas fisheries, *etc.*), this will imply the need for significant international co-ordination.

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