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Assessing the Economic Impacts of Soil Degradation

Final Report

Volume III: Empirical Estimation of the Impacts

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Géosciences pour une Terre durable

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List of Abbreviations

CAP	Common Agricultural Policy
DC	Defensive Cost (off-site)
ECB	European Central Bank
EEA	European Environment Agency
€ / ha*y	Euro per hectare per year
$€_{2003}$	Euro value, expressed in 2003 prices
FAO	UN Food and Agriculture Organisation
MC	Mitigation Cost (on-site)
MS	Member State
NPV	Net Present Value
NC	Non-user Cost (off-site)
OM	Organic Matter
PC	Private Cost (on-site)
SC	Social Cost (off-site)
SOM	Soil Organic Matter
TEV	Total economic value
TG	Task Group
WFD	Water Framework Directive
WG	Working Group

1 Introduction

1.1 Scope of the Study

In recent years, there has been growing concern about the quality of European soils. Intensive agriculture, increasing settlement and urban sprawl as well as climate change are affecting soils across Europe. Whereas soil in its original state is able to perform a multitude of functions, it looses this capacity under pressure. This process is described as *soil deterioration*, and occurs both naturally and as a consequence of human influences. This study focuses only on *soil degradation*, a term that is used to describe man-made impacts on the different soil functions. It does not consider natural soil deterioration processes that occur without man-made contribution.

Soil degradation has been assessed from different angles, however so far little comprehensive research has been done to estimate the economic damage caused by soil degradation. The present study aims at assessing the economic impacts of soil degradation in Europe. In doing this, the study has looked at the **costs of non-action**: what are the costs that are presently caused by the failure to use European soils in a sustainable way? It can be assumed that these costs of non-action are approximately equal to the **benefits of action**: if it was possible to put an end to soil degradation through appropriate measures, most of the costs would no longer be incurred. However, this study did not assess the **costs of action**: the costs of applying a certain policy that would limit or eliminate soil degradation, e.g. by changing agricultural and land use practices.[1] Neither did this study assess the **benefits of non-action** – which consist not only of the saved cost of action, but also comprises the short-term benefits that soil users derive from degrading soil use practices (see also Box 1).

In order to assess the economic impacts of soil degradation, this study has used a methodology that was developed based on a review of the existing literature on the topic. This literature review (Volume I of this report) showed that some empirical studies have tried to assess the economic impacts of soil degradation in monetary terms in recent years. However, the majority of these studies was conducted in Northern America and Australia. European studies, by contrast, are few and far between: the literature review identified twelve European studies with quantified economic information. In addition, most of the empirical estimates were derived on a local basis only. In Europe, few studies have extrapolated the impacts of soil degradation beyond the immediate study area to a regional or national scale.

The few studies that have tried to provide countrywide estimates of the costs of soil degradation, typically rely on a number of bold assumptions. This is especially true of studies conducted at the EU level (e.g., Van den Born et al. 2000) or for Asian countries (Ahmad and Kutscher 1992, FAO 1994), where the data availability is limited. To a lesser degree, this is also the case for studies that describe the situation in North America (e.g., Ribaudo 1989, Science Council of Canada 1986, Den Biggelaar et al. 2001, Clark, Haverkamp and

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[1] Among the different types of impacts (described in greater detail in chapter 2.1 below), the category of mitigation costs and defensive expenditure are somewhat ambiguous. They are both damage avoidance costs: they capture the cost of measures that do not address the problem of soil degradation as such, but rather the cost of defensive measures that will limit or compensate the impacts. Examples would be fertiliser applications to compensate falling yield levels, or slurry walls that limit the spread of soil contamination. Since these measures do not solve the problem as such, they should be regarded as part of the costs of non-action.

Chapman 1985). Consequently, such nation-wide estimates have been subject to much criticism and debate.

Apart from the general shortage of reliable data when it comes to estimating economic impacts of soil degradation, the literature survey has also documented several other shortcomings that should be borne in mind:

- A considerable amount of research has focused on specific aspects and particular types of soil degradation, and has assessed these soil degradation processes from a natural science perspective. However, economic impacts of soil degradation are generally much less researched. Some types of soil degradation – especially loss of soil organic matter or soil sealing – are partly understood from a natural science perspective, but have not been investigated yet from an economic point of view. For other soil degradation types, especially loss of soil biodiversity, the understanding is still limited even from a natural sciences perspective, which means that an economic assessment is even more difficult. Given the limited availability of empirical data, quantitative statements at this stage are possible for soil erosion, soil contamination and – within limits – for soil salinisation. These three soil threats will therefore be considered in greater detail in this study.
- Of the economic data that is available, most does not come from Europe, but from studies that have been conducted abroad – mainly in Australia and North America. If anywhere, a conclusive interpretation of the economic impacts of soil degradation would be possible in the US. For Europe, the economic evidence is limited. Therefore, while a description of economic impacts is possible, it is not possible to quantify all of these.
- The surveyed studies have also tended to focus on the on-site impacts of soil degradation. This is especially true for the types of soil degradation that are associated with agriculture, such as erosion and salinisation. Here, research has mainly focused on agricultural productivity impacts. However, the focus on on-site impacts found in many (agronomic) studies is misleading, as these often represent only a minor part of the total economic damage. The greater part takes the form of off-site impacts, e.g. through siltation and sedimentation caused by erosion, but also through the impacts on ecosystem services provided by soils.[2] However, off-site impacts are more difficult to identify and measure. This is partly because the damage caused is more diffuse – in the sense that it is spread over a very large number of actors who are not necessarily aware of the welfare loss they incur. As a consequence, there is much less empirical evidence on off-site impacts of soil degradation.

Furthermore, there are two types of impacts (both subsets of the off-site effects) for which empirical evidence is extremely scarce or non-existent.

- This firstly concerns the non-use values of soils, i.e. the values attached to soil by people who are not currently using it, nor intend to used it in the future (see also Box 6 on page 33). Such non-use values reflect the conviction that soil should be preserved as a resource in its own right (existence value) or for use by future generations (patrimonial or bequest value). They can also reflect the cultural, spiritual or religious value of soils. Studies that conducted for other environmental goods suggest that non-use values can account for a substantial share of the Total Economic Value of environmental goods (see e.g. Görlach and Interwies 2003 for the case of groundwater). The fact that these values

[2] See also the case studies on erosion documented in Volume II of this report.

largely escape an economic assessment is mainly due to a lack of data, but also point to a more general confinement of economic valuation methods. These methods are more suited to describing environmental goods where they produce direct, measurable economic values, but less so where the economic value is due to individual beliefs and convictions. The lack of economic information in the case of soil means that the economic assessment of soil degradation is largely confined to the use value of soil, i.e. the human uses of soil that are of economic relevance.

- The second category where data is severely deficient concerns the ecosystem services provided by soils. The concept of ecosystems services reflects the fact that the parts of the ecosphere are closely interconnected. In this way, soil degradation will also affect the hydrosphere and the atmosphere, as buffering and filtering functions of soils are lost.

Despite these limitations, the present study attempts to offer a valuation of the economic impacts of soil degradation for the European Union. It should be clear that the results of this study have to be interpreted with caution. The purpose of these calculations is not so much to come up with one definite and exact number for the cost of soil degradation in Europe, but rather to give an idea of the scale of economic damage and to define further research needs.

In order to deal with the shortage of empirical data, this study has to rely on a number of assumptions. Care was taken to make these assumptions explicit throughout, as well as explaining the motivation for why they were made. Despite the assumptions made, it has not always been possible to arrive at monetary estimates for the economic impacts of soil degradation. Therefore, results will not only be presented in monetary form, but also in other quantified forms, or as qualitative information where no other information was available.

1.2 Outline of the Study

Following the introduction, chapter 2 describes the methodology that was applied in this study. Chapter 3 briefly summarises the data sources that were used for the extrapolation. The extrapolation results themselves are presented in chapter 4.

Since the previous economic research is scarce for many types of soil degradation, it was agreed in that this study should focus on those soil threats where most data is available. Three types of soil degradation are assessed in greater detail, namely

- Soil erosion,
- Soil contamination, and
- Soil salinisation.

For these three threats, monetary estimates of the economic impacts are presented. For the other soil threats that identified in the European Commission's framework, a quantitative or qualitative assessment is provided, incorporating information in monetary form where possible. These threats are discussed in Annex 1 of this document.

This document forms part of the final report for the project "Assessing economic impacts of soil deterioration", which was commissioned by the European Commission, DG Environment, to Ecologic and French Geological Survey BRGM (study contract ENV.B.1/ETU/2003/0024). It is supported by, and builds on, an extensive review of the relevant literature in this field (Volume I of this report). It also builds on the results of a number of case studies that have been assembled for this project, and on a Database research carried out by BRGM (Volume II of this report). This document is a joint product by Ecologic and BRGM, where BRGM has the main responsibility for the subchapter on the costs of contamination.

2 Methodology for the Assessment

The methodology for this assessment was developed from similar approaches used in comparable studies. The literature review that was carried out in support of this project explains the methodology in greater detail (Volume I of this report, chapter 4.6).

2.1 The Methodology derived from the Literature

The approach that was applied in this study aims to achieve a justifiable trade-off between the complexity that is inherent to soil degradation and its impacts, and the scarcity of economic data that is available to measure this complexity. From the literature review, it has become clear that there are several aspects of soil degradation that are relevant to an analysis of the problem, but are difficult to assess in monetary terms. These aspects include the multifunctionality of soil as well as the multitude of soil types and soil uses that can be affected by soil degradation. Other aspects that are difficult to quantify include the value that soil has independently of (human) uses, such as ecosystem services provided by soil, and patrimonial values attached to it, such as spiritual or cultural connotations. In order to deal with these complexities, workable and pragmatic solutions have to be found, including a number of assumptions to reduce complexity. In such cases where even heroic assumptions would not allow quantifying the impacts, results are discussed in a qualitative way instead.

One key component of the proposed methodology is the distinction between five different cost categories.[3] They include:

- The **on-site (private) costs of damage** suffered as a consequence of soil degradation. An example for this is the yield loss that farmers incur if the agricultural productivity of soil has been reduced through erosion, compaction or other degradation processes. These costs are denoted **PC**;
- The **on-site private cost of mitigation and repair measures** to limit the impact of degradation or to prevent further degradation. This includes, for example, the cost of additional fertiliser input to compensate for the impact of erosion, or the cost of measures to restore the physical structure of compacted soils. This category is labelled **MC**;
- The **off-site (social) costs** of soil degradation, which are suffered by other parties. One example is the cost of damages caused by floods and landslides. It also includes the value of foregone ecosystem services, such as biodiversity maintenance or carbon sequestration, which are reduced through soil degradation. These costs are denoted **SC**;
- The **off-site defensive costs** incurred in order to mitigate or limit the off-site impacts of soil degradation. This includes e.g. the cost of soil conservation measures to prevent landslides, or to retain the soil on the site. These costs are abbreviated as **DC**.

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[3] The methodology was developed based on a review of the literature. It combines different categorisations that can be found in the literature: first, the distinction between on-site and off-site effects (also referred to as on-farm and off-farm effects) is common in economic studies on soil degradation, see e.g. Huszar and Piper 1986, Brouwer et al. 2002, Prosser et al. 2003, Boardman et al. 2003, Ribaudo 1989. Secondly, the distinction between use values and non-use values is routinely applied in environmental economics, see e.g. Pearce and Howarth 2000, Bonnieux et al. 1998, Turner et al. 2003. The distinction between cost of suffered damage (damage cost) and cost of defensive measures (damage avoidance cost) is explained by Turner et al. 2003b and DG Eco II 2004.

- The **non-user costs** that accrue to the individuals that do not use the soil, but are nonetheless distressed by its degradation. This category measures the non-use values attached to soil, e.g. the patrimonial value of preserving soil for future generations. Where such values are affected by soil degradation, the cost are captured as **NC**;

The definition and delineation of the different cost categories, as well as examples for each category, can be found in the literature review (Volume I of this report). Figure 1 gives a schematic overview of the different cost categories. As depicted in the figure below, the two categories PC+MC together constitute the on-site costs of soil degradation, while the total off-site costs are calculated as the sum of SC, DC and NC.

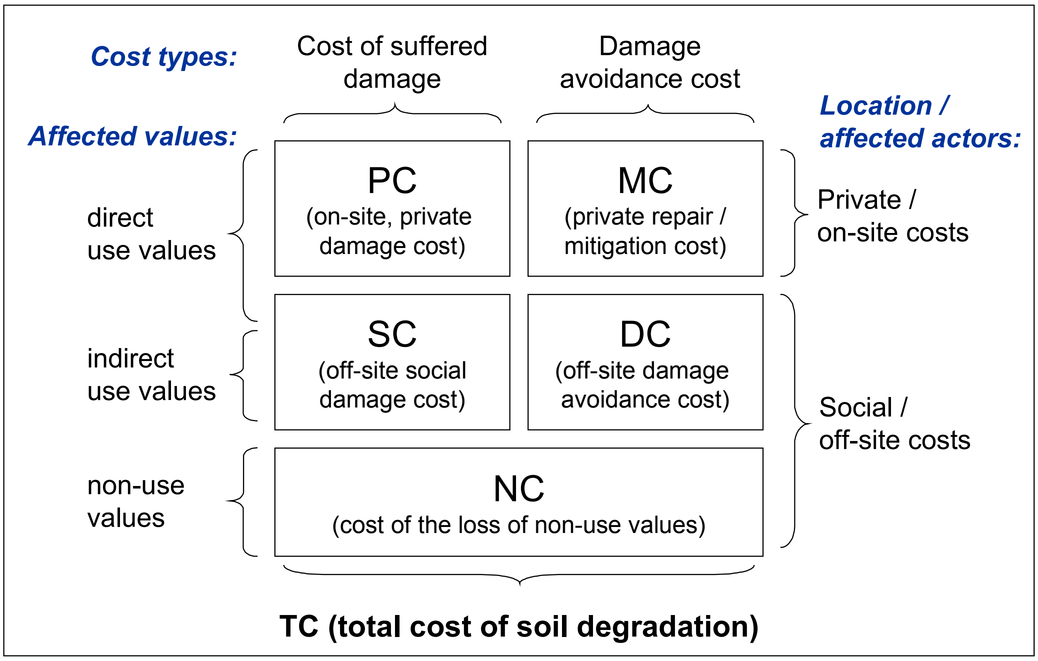

Figure 1: Overview of the different cost categories

A word of explanation is necessary for the relation between the cost of suffered damage and damage avoidance costs (left-hand-side and right-hand-side in the figure above). The focus of this study is to assess the costs of non-action, i.e. the economic damage caused by the unsustainable use of soils. Therefore, the cost of suffered damage would appear more relevant for this study. However, there are two reasons why damage avoidance cost also need to be considered:

- The first is a methodological reason: not all of the damage caused by soil degradation will occur unmitigated. Through mitigation measures, the impact of soil degradation can be partly offset (e.g. by applying fertilisers to compensate for falling yields). However, such measures do not address the problem itself, but rather cure the symptoms. In this sense, they need to be included among the cost of *non-action*: if soils were used and managed in a more sustainable way, mitigation measures would no longer be necessary.
- The second reason is empirical: the costs of suffered damage may be difficulty to assess in some instances – especially for (off-site) social costs. Here, damage avoidance costs can provide an alternative. This approach assumes that the cost of repairing, restoring or remediating a deteriorated environmental asset can be taken as a (lower-bound) proxy of the damage cost caused by its degradation (Turner et al. 2003b). Damage avoidance costs will therefore be used for threats where parts of the damage cost caused by soil degradation cannot be assessed (see salinisation or contamination).

For both types of damage avoidance costs (MC and DC), it should be noted that they do not indicate the cost of a full restoration to pristine conditions (which is impossible in most instances). Rather, they reflect the cost of measures that are taken to limit or mitigate the impact of soil degradation, e.g. limiting the impact of erosion on yields through fertiliser, or returning contaminated land to a “fit-for-use”-state on a risk assessment basis, as commonly agreed in European Countries for the management of contaminated sites and soils.

For the category of social costs (SC), it is important to underline that the categorisation is primarily an economic one. In this sense, social costs are costs that are borne by society – in opposition to private costs, which are covered by the soil user / owner. Therefore, the category of social costs does not address explicitly impacts that would commonly be regarded as “social” in nature, such as the cost of unemployment, or the economic consequences of land abandonment and rural depopulation.

It is important to realise that the role of the five cost categories differs between soil threats. This is apparent for the category of on-site mitigation costs: For the case of contamination, this category is much more relevant than e.g. for erosion or salinisation, since more efforts are undertaken and more funds are assigned to the remediation of contaminated land. Also, the quality and nature of the mitigation cost data differ for the different soil threats: for contamination, the mitigation costs are based on actual expenditure per country, which is fairly well documented e.g. by the European Environment Agency. In other cases such as salinisation or erosion, mitigation costs were calculated based on the extrapolation of indicative data to the area affected. For these threats, the mitigation costs form a smaller share of the total impact, while the suffered impacts are more relevant.

Box 1: Farm-level Costs and Benefits of Soil Degradation

This study has focused on the costs of soil degradation rather than its benefits. Nonetheless, it is important to realise that land management practices causing soil degradation will often deliver a short-term benefit to the land user. Indeed, this short-term benefit is one of the causes of soil degradation. In the short run, and as long as off-site impacts are not factored in, it may be economically viable for land users to exploit the soil beyond its natural rate of renewal, even though this may lead to permanent degradation. This applies to farmers employing erosive land management practices as well as to firms that economise on protection measures against soil contamination.

In the agronomic literature, the farm-level decision on whether to apply conserving or degrading practices has been covered in some detail (see e.g. Eaton 1996 or Pagiola 1994). The decision can be described as an optimisation problem, where a farmer chooses the land management strategy that maximises yields. Figure 2 below presents the problem in a schematic way (based on Anderson and Thampapillai 1990).

The left panel depicts the maximum attainable yield without conservation measures as well as the yield derived with sustainable soil use. The farmer can chose between these two land use practices: he can either maximise yields without applying conservation measures. This can be pursued for a number of periods without any impact on the yield (point t_1 below). After this point, soil functions are affected as the soil is degraded, consequently yield levels start to decline, until the yield eventually converges to zero (i.e. complete degradation). Alternatively, if the farmer chooses to manage his land in a sustainable way, the yield will be lower initially. However, over time the yield remains stable and may even increase as soil functions are enhanced, e.g. by building up soil organic matter. At a given time (point t_2 below), the sustainable yield will exceed the yield derived from a degrading strategy.

The right panel presents the accumulated yield for the same case (on a different scale). It shows that the accumulated yield initially grows faster for the degrading strategy, but eventually levels off. By contrast, the accumulated **sustainable** yield continues to grow until it surpasses the yield obtained with a degrading strategy (point t_3 below). The point t_3 (where accumulated yields are equal), will be reached later than t_2 (where current yields are equal).

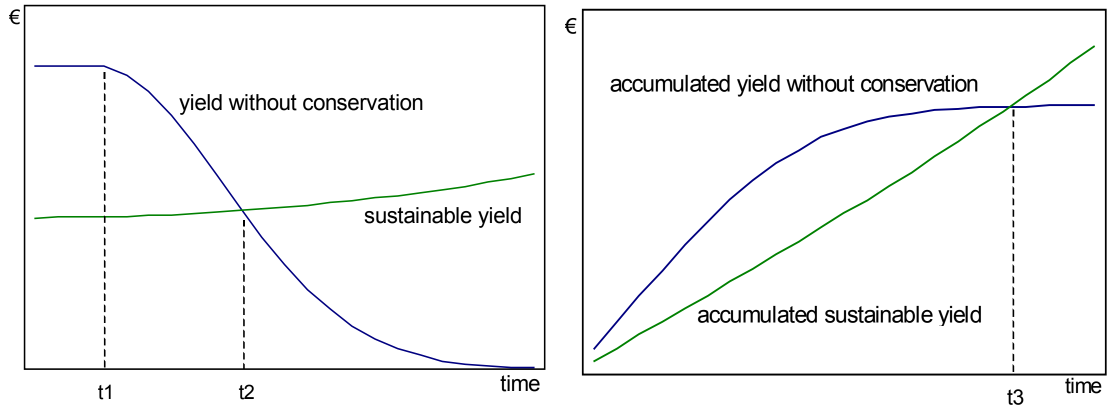

Figure 2: Yields for different soil management strategies

This simple model shows that a sustainable strategy will pay off in the long term, even if it leads to lower yields in the short run. However, there are two main reasons why it may be perfectly rational for a farmer to employ a degrading strategy nonetheless:

- **Length of the planning horizon**. The point t_3, where the sustainable strategy begins to pay off, may be several decades in the future (depending on the initial state of the soil). If the farmer only plans for the next few years, so that the point t_3 is outside his planning horizon, it will be more economical to follow an unsustainable path of action.
- **Discounting**. Through discounting, yields in the future will be valued lower than present yields. This means that the benefits of following a sustainable strategy, which accrue in the long run, will receive less weight than the immediate benefits of the degrading strategy. This has the practical effect of shifting t_3 outward, to a later point in time. If a high discount rate is applied, t_3 might not be reached at all.

Besides these economic arguments, there are also some down-to-earth reasons why farmers may apply a degrading land management practice (see also Eaton 1996):

- **Lack of information**. Farmers may either not perceive soil degradation as a major risk, or they may be unaware of the impacts of their action. And even if they are aware, they may fail to act differently as they lack knowledge about alternative options.
- **Uncertainty about future impacts**. Even if all options are known, it will be difficult to estimate the impact on future yields with the same precision that is suggested by the graphs above. Real-life decisions will involve much more uncertainty.
- **Land ownership**. If a farmer does not own the land he is cultivating, and if the land owner does not have a way of monitoring and influencing land management practices, the farmer has no incentive to preserve the soil beyond the rental period.
- **Agricultural subsidies**. The overall effect of subsidies on land management strategies is ambiguous. On the one hand, subsidy schemes may be explicitly targeted to support sustainable land management practices, e.g. through the CAP cross-compliance

mechanism. On the other hand, subsidies may create an incentive to increase yields in the short run, or they may lead to a false sense of security and disregard for future risks.

- **Availability of credit and funding**. To apply sustainable land management implies lower yields in the initial phase, and may require additional investment for soil conservation measures. If sufficient credit is not available, a farmer may be forced (against better knowledge) to adopt land management practices that result in soil degradation.

It needs to be emphasised that this argumentation only considered the private, on-site impacts of soil degradation, assuming that these are most relevant for the farmer's decision. If off-site impacts were taken into account, they would need to be subtracted from the yield obtained with non-sustainable land management. This means that the net benefits (yields minus off-site impacts) of this strategy would fall much faster over time, and would eventually turn negative. This clearly affects the decision in favour of sustainable land management.

A main reason why agricultural land use may lead to soil degradation is thus that farmers lack adequate incentives to take off-site effects into account. The same is true for long-term impacts that are also not adequately reflected. Several mechanisms can be identified in order to correct these incentives. This can be done through economic instruments (taxes or subsidies), through codes of conduct (good agricultural practice), but also through better information about the long-term impacts of unsustainable land management. The latter may also help to give more weight to patrimonial or bequest values in farm-level decision making. This reflects the growing recognition that agriculture should not only aim to maximise yields, but that is also plays a key role in preserving soil functions and protecting the environment.

2.2 Limitations of the Approach applied to Soil Degradation

Concerning the application of this proposed methodology, several limitations should be noted. These limitations arise both on a theoretical-conceptual and on the empirical level.

2.2.1 Theoretical limitations

On the theoretical or conceptual level, there are several limitations that are either specific to the assessment of soil degradation, or that apply to the economic valuation of environmental resources in general. While some of these limitations can be addressed through specific assumptions, the number of limitations suggests that any estimate of the impacts of soil degradation is likely to be conservative:

- Some costs can only be approached as lower-bound estimates, or cannot be calculated at all. Impacts that are not quantified in physical terms, not sufficiently understood or not even known of do not appear in the calculation (see also Pretty et al. 2000). This may be particularly problematic for ecosystem services provided by soils, as well as the non-use values attached to soil (see below).

- Damage avoidance costs (mitigation costs, MC, and defensive expenditure, DC) are calculated as incremental costs only, necessary to prevent further damage and limit its impacts. They are not measured as the total cost of returning soils and other ecosystems to pristine conditions.[4] The cost of a full restoration would clearly be much higher (see also Box 7).

[4] Damage avoidance costs can be measured as the cost of returning soil to a "fit-for-use" state, falling short of a complete restoration.

- Furthermore, it should be noted that the temporal dimension of soil degradation is difficult to reflect adequately in a large-scale extrapolation. Soil degradation has impacts in the short, medium and long term. Some of the impacts of soil degradation may be irreversible on a human time scale, and impacts may be cumulative and mutually reinforcing (see also chapter 5.5 the UK case study in Volume II of this report). Whereas these different factors can be reflected in a limited, local case study, the current extrapolation was restricted to calculating the annual average costs of soil degradation. Hence the capacity to illustrate long-term effects and irreversible impacts is limited. The economic approach of summing up impacts in the form of a net present value was not followed due to concerns about the choice of interest rate. For a discussion, see Box 2 below.
- One methodological difficulty stems from the fact that the different types of soil degradation will often occur in conjunction and therefore cannot always be strictly separated. For instance, this is often the case for soil erosion, loss of organic matter and loss of soil biodiversity. It also applies to floods and landslides, which are often caused or at least aggravated by soil erosion, sealing and compaction in upstream areas. In the current study, the impacts of a loss of organic matter and of soil biodiversity have not been monetised due to a lack of data. However, it appears safe to assume that the estimated costs of erosion partly capture the impacts of organic matter and soil biodiversity loss. Unfortunately, this relation could not be specified further.
- It should be noted that the empirical estimates included in the literature survey are generally based on a partial-equilibrium approach, meaning that they calculate the impact on the study site only, and assume that other factors (such as the prices of production factors and of agricultural output) remain equal. For a European-wide extrapolation of the impacts of soil degradation, it would be appropriate to incorporate these factors into the modelling. This would, however, reach beyond the scope of the current project.
- Many impacts of soil degradation are closely related to agricultural uses of soil. However, the agricultural sector does not easily lend itself to an economic analysis, as it is strongly influenced by agricultural subsidies. This means that farmers' choices, such as which crop to grow when, how and where, will be influenced much more by the allocation of subsidies than by soil conservation requirements. Furthermore, it is difficult to assess whether the lost agricultural income due to soil degradation should include the foregone subsidy payments, or whether yield losses should be valued at net prices.[5]
- Finally, it has to be noted that the extrapolation does makes use of values inferred from different (spatial or temporal) contexts. However, it is no *benefit transfer* in the strict sense, as this would require a better adaptation of the primary data to the cases to which it will be applied. For example, it will be assumed that the impact of soil salinisation or erosion on agricultural productivity is approximately equal across different countries. Also, in transferring the off-site costs of soil erosion to other sites, the calculations did not account for differences in population density or downstream water uses.

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[5] In this study, foregone subsidies were generally not included in the production losses suffered by farmers; instead, market prices were applied (cf. also the case study on salinisation in Volume II of this report). A different situation arises where public support is paid for soil conservation measures, these costs were included either as defensive expenditures (DC) or social costs (SC).

Box 2: Intertemporal Valuation of Soil Degradation

The practice of economic valuation as applied in this study allows estimating the economic impacts of soil degradation at a given point in time. Yet, as soil is a non-renewable resource, the economic impact of soil degradation will be felt for several decades or even centuries, in the same way that we currently feel the impacts of soil degradation in the past. In other words, the impacts are cumulative. A full assessment should therefore to consider not only the current impact of soil degradation, but also medium and longer term impacts.

The standard economic approach to dealing with costs and benefits that accrue in the future is to discount them: based on the assumption that individuals value costs and benefits in the present higher than future costs and benefits, the latter are divided by a discount factor. This allows calculating the *net present value* of the current and future impacts of soil degradation:

$$NPV = \sum_{t=1}^{T} \frac{C_t}{(1+r)^t}$$

where
- t indicates the time period;
- T indicates the time horizon that is considered;
- C_t indicates the cost of soil degradation in the period t; and
- r is the discount rate that is applied.

In the literature, there is some discussion about the appropriate discount rate to be applied, and the time period to be considered. Standard discount rates for the valuation of natural resources and conservation projects normally range from 3 to 6 percent (ASTSWMO 1998). However, some experts argue that discounting should not be applied at all, as the idea seems debatable that natural resources preserved for the future should be valued lower than resources available today.

Along these lines, Young (1998) argues that standard discounting procedures are not applicable to soil, since they implicitly assume a substitutability of resources. If discounting is applied in the intertemporal valuation of soil, this assumes that future technologies will allow a more productive use of the remaining soil, or a substitution of soil with man-made capital. Young denies that this is the case, since there may not be any substitutes for degraded soils. Rather, if production shifts to marginal lands, this will lead to even more and accelerated degradation, requiring higher inputs of labour, fertiliser and machinery to deliver the same yields. As a consequence, Young argues that a discount rate of zero should be applied to the valuation of soil degradation over a period of 500 years. This would imply that future losses are balanced directly against the current costs, without any discounting.

In this study, the issue of discounting was dealt with by considering only the current, annual cost of soil degradation. This does not fully avoid the question of discounting, because the costs of longer-term measures still have to be annualised; however, it circumvents the choice of a discount rate for future benefits of soil protection. The costs per annum can then be combined with a qualitative assessment of how the costs are expected to develop over time, including an assessment of the uncertainty associated with this forecast. The trade-off of current benefits against future costs is then left to the audience, and can be decided e.g. in consultation with relevant stakeholders.

In order to illustrate the impact that the choice of the discount rate has on the result of the analysis, some exemplary calulations are presented in section 5.3.

2.2.2 Empirical limitations

It is evident from the literature that on-site effects of soil degradation have received far more attention than the off-site effects. Within the assessment of on-site effects, there is a strong focus on agricultural causes and impacts.

- On-site effects have mainly been the subject of agronomic research, especially for agriculture-induced cases of erosion and partly for salinisation, focussing on the link between soil degradation and agricultural productivity. Also, soil degradation causes other than agricultural practices and impacts on other sectors (e.g. tourism) are considered much less.
- Despite recently increased interest in the subject, the off-site effects of soil degradation are generally less researched. From the existing literature, it is widely acknowledged that the off-site effects form a significant part of the total cost of soil degradation, and will often exceed the on-site costs (see e.g., Clark et al. (1985), Crosson and Stout (1983), Crosson (1986), FAO (1999), Furtan and Hosseini (1997) and Pretty et al. (2000). For the case of erosion in the UK, this view is strongly supported the UK case study in Volume II of this report.

Other impact categories with even less data are the ecosystem services, which are included in the SC category above, or the non-use values, represented as NC in the above system of cost categories:

- Ecosystem services are the services that soil provides in interaction with other ecosystems, including the regulation of the natural water cycle, nutrient cycling, the creation and absorption of biomass, the sustenance of biodiversity, and the natural carbon, sulphur and nitrogen cycles. These soil functions are of enormous importance for human survival and for economic activity. However, so far, their economic value has not been assessed comprehensively. Neither has it been assessed how the economic value of ecosystem services is affected by soil degradation.
- The category of non-use values is still more evasive. Non-use values are negatively affected if someone who is neither currently using the soil, nor intends to do so in the future, experiences soil degradation as a loss. Non-use values can take the form of existence values, based on the conviction that soil should be protected in its own right, or they can take the form of bequest values, if the soil is preserved for future generations. In this sense, the non-use value can also capture spiritual and patrimonial connotations associated with soil. However, empirical evidence on non-use values is virtually non-existent, safe for a few Australian studies (cf. Box 6 and chapter 4.1.3).

3 Data sources for the Assessment

3.1 Soil Data

The data sources for the soil-specific data that were used for this extrapolation are described in detail in the volume "Case Studies and Database Research" (Volume II of this report) that was produced as part of the project "Assessing Economic Impacts of Soil Degradation". Chapter 4 of the document contains an assessment of the availability, reliability and spatial coverage of the data sources that were used for this extrapolation.

3.2 Economic Data

Two types of economic data sources are used in the extrapolation: (i) case studies conducted as part of this project and (ii) information found in the literature (Volume I of this report). The literature survey covered economic, agronomic and other relevant economic journals, and furthermore included contacts with leading experts in the field in order to assess the availability of data and literature, and to avail of unpublished literature.[6] Next to literature from peer-reviewed journals, evidence was also taken from government- or EU-funded research projects. The surveyed literature included, inter alia, about 60 empirical economic studies that have quantified the economic impacts of soil degradation for different soil threats.

The economic data that was derived from the literature is generally reported in Euro values, which are discounted to 2003 using the ECB's annual inflation figures. Estimates from outside Europe were adapted based on purchase power parity values. In most instances, costs are reported as annual costs.

In addition to the economic data gathered from the literature survey, statistical data was also used for the extrapolation on several occasions, such as agricultural statistics provided by DG Agriculture and by Eurostat (European Commission 2001, Eurostat 2002).

[6] The experts that were contacted in the course of this study include Rob Evans of the Anglia Polytechnic University; Tore Söderqvist of the Beijer Institute, Stockholm; Reinhard Schmidtke of the Bavarian Water Management Agency, Munich; John Boardman, University of Oxford; Anthony Young, University of East Anglia; David Dent, ISRIC; Andreas Bieber of the German Federal Ministry of the Environment; and Martin Socher of the Saxonian State Ministry for the Environment and Agriculture. The authors would like to thank them for their valuable support. This study has also benefited from discussions at the Vital Soil Conference that was held on 18 and 19 November 2004 in Scheveningen, NL, as a joint conference by the Dutch EU Presidency and the European Commission.

4 Economic Assessment of Different Soil Degradation Types

4.1 Erosion

4.1.1 Situation

According to EEA (2003c), soil erosion, is one of the major and most widespread forms of land degradation in Europe, and poses severe limitations to sustainable agricultural land use. EEA (2000c, 2003c) notes that erosion takes different forms in the different parts of Europe. Soil erosion by water and wind is most severe in Southern Europe, where it is a well-recognised problem. Southern European countries are most prone to severe water erosion: long dry periods are followed by heavy bursts of intensive rainfall, falling on steep slopes with fragile soils and low vegetation cover (EEA 2003c). The most severe soil losses are observed in the Mediterranean (Southern Spain, Southern France, Italy and Greece), the Balkan Peninsula and the countries surrounding the Black Sea (EEA 2000c). In some of these regions, erosion has reached become irreversible to the point that no more soil is left. Especially in Southern France and Southern Spain, occasional storms may lead to erosion of 20 – 40 t/ha, with extreme events leading to losses of up to 100 t/ha (EEA 2003c).

Wind and water erosion can also be observed in large parts of central and eastern Europe, where the impacts of erosion are often exacerbated by historical soil contamination. Moderate rates of water erosion can be found in loess soils in the North of Europe. Although less severe than in Southern Europe, erosion is becoming increasingly problematic in the northern European loess belt. Erosion in these latitudes mostly results from less intense rainfall falling on saturated, easily erodible soils, especially where these are subject to intensive agriculture. There is also local wind erosion of light soil. Erosion rates in the Northern European loess belt tend to be moderate at up to 10 t/ha*yr (cf. EEA 2000c, Volume I and Volume II of this report).

4.1.2 Method

4.1.2.1 Distinction of Impacts and Cost Categories

The following section discusses how the cost categories identified in chapter 2.1 can be applied to the case of erosion. For a description of the different cost categories, please refer to chapters 2.1 and 2.2).

The direct impacts of soil erosion are mainly losses of soil. These are measured in tons per hectare per year (t/ha p.a.). Erosion also reduces the fertility and productivity of soil. One example is the removal of plant nutrients and organic matter, or the decrease in plant rooting depth. Soil losses can also lead to uprooting of plants and trees. These effects can reduce yields and hence affect agricultural productivity. But they may also have an impact on tourism if they occur on a larger scale. These impacts of erosion constitute the **private costs of erosion (PC)**.

To compensate for erosion-induced yield losses, land users have to apply more fertilisers to degraded soils, or apply stabilisation and conservation measures to prevent further erosion. This part of the impacts is labelled as the **mitigation costs (MC)**. It has to be noted that the mitigation measures will only cure the symptoms, but not address the problem itself. Applying more fertilisers may keep yield levels constant for some time, but will not eliminate the underlying root causes for erosion. Mitigation costs are therefore included among the costs of non-action (see also chapters 1.1, 2.1 and 5.1 for a discussion).

These two categories, private costs (PC) and mitigation costs (MC), together form the on-site costs of soil erosion. In addition, erosion causes considerable economic damage outside the area where erosion actually takes place. This damage is referred to as off-site impacts.

Off-site impacts are mainly related to the environmental functions of soils. They include damages to natural ecosystems as well as to economic uses of these ecosystems. Most off-site impacts are transmitted through the water cycle. This can either occur as physical damage to water bodies (siltation of dams, sedimentation of rivers and canals), or as chemical damages (in many parts of Europe, soil erosion is a primary source of diffuse water pollution). As erosion increases sediment loads in water bodies, it forces water utilities to invest in water treatment or local defence measures (for instance retention and sedimentation pools). Similarly, sediment loads have a negative impact on natural habitats and on fisheries, affecting commercial and leisure activities. Since erosion weakens the water-holding capacity of the soil, it may increase the risk of floods and landslides.

Depending on whether measures are being taken to address these off-site impacts, the off-site costs of erosion are either classified as **social costs (SC)**, measuring the cost of suffered damages, or as **defensive costs (DC)**, the cost of defending against further damage. While the treatment cost for water affected by sediment is an example of suffered damages (hence SC), the cost of retention ponds would fall under the measures to defend against further damage (hence DC).

A fifth category captures the **non-use value of soil (NC)**. These could not be quantified in this study, as there is virtually no information. Non-use values includes e.g. the cultural and spiritual functions that soil has as an archive of human history, or the values that people attach to a specific landscape, to the extent that these are affected by erosion.

Table 1: Impacts of Erosion sorted into Cost Categories

On-site costs	PC	*Example*: Yield losses from eroded agricultural soils, impact on tourism
	MC	*Example:* Costs of stabilisation and conservation measures (hedges, etc.)
Off-site costs	SC	*Example:* Costs of siltation of dams and canals (commercial and recreational uses) and costs due to increased sediment load for surface water users
	DC	*Example*: Costs to prevent or clean up sedimentation of dams and canals
	NC	*Example*: Impacts on landscape values and biodiversity

Box 3: The Impact of Erosion on Ecological Functions of Soil

Soil provides a multitude of functions. Fertile soils are an essential component of global ecosystems, as all plant and animal species need either soil itself or products that are grown in soil in order to survive. Erosion affects these soil functions to a different degree. The affected soil functions include (WG on Erosion, TG 3 2004):

- food and other biomass production;
- storing, filtering and transformation of minerals, organic matter, water, energy, and chemical substances, including the role of soil in the global water and carbon cycles, and
- the function of soil as habitat and gene pool.

However, only few of these functions are of direct and measurable economic relevance, e.g. for agriculture or tourism. In other cases, their valuation may be extremely difficult if soil functions do not produce any measurable economic benefit, or where their loss does not lead to a measurable economic damage.

From an economic perspective, two ways of valuing these soil functions (or their loss) are feasible in principle: either through non-use values attached to soil and its functions, or through the concept of ecosystem services.

- Non-use values are present where someone who is not using the soil, nor intends to use it in the future, is prepared to contribute to its improvement or conservation. Non-use values are typically measured through interviews that elicit people's willingness to pay.
- A second approach is to value ecosystem services, i.e. to quantify in monetary terms the numerous services that ecosystems provide. A theoretical discussion of the concept can be found in Volume I of this study, whereas Box 8 presents some quantitative evidence.

With regard to non-use values, Box 6 demonstrates that the data situation for assessing them is extremely poor in Europe. Regarding ecosystem services, the data situation is somewhat better, but still far from satisfactory. In particular, the limited evidence does not allow to assess the relative importance of different soil functions in a methodologically consistent way. Crosson (2003, p.12) states that "economists have only within the last few years begun to work on soil quality and carbon sequestration issues, and much of that work is still not published. … What one can plausibly say about the economics of maintaining or enhancing soil biodiversity thus is very limited."

Despite these limitations, some quantitative evidence are presented in this chapter (see also Table 7), such as:

- The function of soil as a carbon pool (and subsequent climate change impacts if this function is affected). Valued at € 60.12 /ha*y, this cost represents 35.6% of the upper-bound estimate for the social cost (SC) category;
- The function of soil for regulating the water cycle and holding back run-off after rainfall (and subsequent impact on floods if this function is affected). At € 8.49 /ha*y, this represents 10% (5%) of the intermediate (upper-bound) estimate for the social cost (SC) category;
- The function of soil as part and basis of the natural landscape (and subsequent impact on recreational uses if this function is affected). Valued at € 19.78 /ha*y, this represents 23% (11.7%) of the intermediate (upper-bound) estimate for the social cost (SC) category.

Regarding the function of soil as a habitat and gene pool, Pimentel (1997) investigates the economic value of activities that take place in the natural soil biota. Such combined activities aerate the soil, facilitate the formation of topsoil and increase the rates of water infiltration, and thereby also enhance plant productivity. Pimentel estimates that earthworms and other invertebrate species bring between 10 and 500 t/ha*yr of subsurface soil to the surface, where the presence of soil biota aids the formation of approximately 1 t/ha*yr of topsoil. Based on this assumption, Pimentel applies a value US$$_{1995}$ 12 (€$_{2003}$ 10.80) per ton of topsoil. If these values, which Pimentel considers as conservative, are transferred to a European context, the total value of soil biota activity to soil formation on agricultural land in the EU-25 (approximately 170 million ha) would then amount to approximately €$_{2003}$ 1.8 billion per year. However, this only represents a total value – it is not possible to establish how this value is affected by soil degradation, or how much of the total is lost every year.

Box 4: Comparison with the Results of the WG on Erosion, TG 3

In its final report dated May 2004, the Working Group on Erosion, Task Group III on Impacts of Soil Erosion addressed a.o. the economic impacts of soil erosion. However, the report focuses on the methodology of assessing economic impacts, explaining how different impacts could be measured or described, and citing selected pieced of quantified evidence. The report does not offer a categorisation or a definite approach. It does underline the difficulty of coming to definite results or drawing conclusions. The report is rather sceptical about the possibility of assessing the economic impacts of erosion in Europe (p. 4): “There are no comprehensive, Europe-wide studies of the economic impact of erosion and available data suggest this is a major challenge. A detailed study of the economic impact of erosion at a European scale can probably only be done by collecting data obtained by local or regional studies, that are carried out by regional or provincial authorities, sometimes even at local community level.”

The Task Group report does not specify a particular approach to categorise the economic impacts of soil degradation. It does, however, refer to categorisations that have been put forward in the literature. The distinction between on-site and off-site impacts is mentioned repeatedly, citing different studies’ findings that off-site impacts can be much more severe.

For example, the report cites Verstraeten and Poesen (1999), who have further divided the off-site costs into direct and indirect costs. Direct costs include costs such as the cleaning up of road infrastructures, the repairing of damaged sewage pipes or the damages to private properties. Indirect costs include the construction and maintenance of retention ponds. The division suggests that direct off-site costs in the Verstraeten and Poesen study correspond to the Social Cost (SC) category in this report, whereas the indirect off-site costs correspond to the defensive expenditure (DC) category in this report.

On another occasion, the report refers to Clark et al. (1985), who found that „Farmers close down their farm if the surroundings are getting severely degraded and unemployment rates are increasing. Often many other driving forces are playing here at the same time, *so it is difficult to estimate the social costs caused by erosion*” (p. 14, emphasis added). In this context, the category of social costs appears to be related to effects such as depopulation; however this categorisation is not defined or specified further in the document.

Since the Task Group report does not specify a particular categorisation for the economic impacts of soil degradation, it is difficult to judge whether the categorisation applied in this report is compatible with the results of the Task Group. However, as the categorisation in this report has been developed based on the same literature, and as crucial distinctions such as that between on- and off-site effects are considered throughout, both are compatible.

4.1.2.2 Data situation for assessing the Impacts of Erosion

In the economic and agronomic literature, there is much evidence of the impacts of erosion on agricultural productivity and yields (the PC category). Recently, a number of studies have also assessed the off-site costs of erosion. Most of these have found that off-site costs can be substantially higher than the on-site costs. However, the aggregation of data from different studies is difficult for several reasons:

- In some studies, **the costs of suffered impacts and the costs of measures to prevent them are treated together** as the cost of erosion (PC and MC are presented as a sum);
- **Many studies do not clearly state which erosion estimations were used to calculate the economic impacts**. Only few studies report the costs of erosion for different,

specified erosion intensities. In addition, where studies from different countries or different continents are combined, the definition of severe, moderate and light erosion often may be different.[7] This is especially problematic for countrywide estimations, where the data base is heterogeneous and often not well documented.

- **In the case of off-site damages, several types of damage can be considered** (including sedimentation of dams and navigable rivers or canals, impacts on water quality, impacts on floods and impacts on recreational uses). These different damage types are not always clearly delineated and may sometimes overlap. Also, not all studies address all of these types of damages.

Partly as a result, the estimates for the cost of erosion differ markedly within the different categories. **For the yield losses associated with erosion (PC), cost estimations range from € 0.45 / ha*y for the US** (average value for affected areas, den Biggelar 2001) **up to € 61 for high-risk areas in England** (Riksen and De Graaff 2001). Although both estimates are recent and quoted from peer-reviewed journals, they differ by a factor of 135. It should be noted that the present study used a range of estimates, combining European, North American and Australian evidence. In combining these, differences in agricultural productivity and unit labour costs between the countries were not accounted for.[8]

To deal with these problems, a **pragmatic approach** for the extrapolation was needed. To this end, the empirical estimates derived from the literature were sorted into the different cost categories (PC, MC, SC and DC, data and methods for the different cost categories are explained below and in chapter 2.1). Since all estimates were published in peer-reviewed journals or in government-funded research projects, none of them was rejected on the grounds of lacking scientific quality. Instead, the result were adjusted in the following way:

- First, all values reported in the literature were combined to give an **unadjusted average** (calculated as the arithmetic mean). This represents the upper-bound estimate.
- In a second step, lowest and highest outlier values were excluded from the set (e.g. values related to extremely high rates of erosion). For the remaining values, the arithmetic mean was calculated. This represents the best-guess **mean value**.
- Finally, a **lower-bound estimate** was calculated for each category. This was either done by using the lowest estimate, or by calculating an average of the lowest values in each category. For MC and DC, the lower bound was set equal to zero. This was done for two reasons: first, because of the weak data for the MC and DC category, and secondly because of methodological difficulties of adding up MC+PC and DC+SC.[9]

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[7] Xu and Prato (1995) are one exception; they specify the cost of erosion for intensities ranging from 2-5 tons / ha. Den Biggelaar et al. (2001) document thoroughly the data sources used, and specify cost for different soil types and crop types. However, their categorisation of erosion intensities differs from the one applied here: they group estimates as 1-5 / 5-10 / 10-15 or more than 15 t/ha*yr.

[8] The estimations rely on overseas data because far more economic data on the impacts of erosion is available in North America and Australia. While the transfer of overseas data to Europe is not unproblematic, it should also be noted that Europe itself is fairly heterogeneous, e.g. in terms of soil types, erosion intensities, and agricultural practices. In this sense, the differences between the impact of erosion in Europe and North America may be comparable to the differences found within Europe.

[9] Adding up the on-site costs of suffered impacts (PC) and the on-site costs of mitigation and repair measures (MC) involves a danger of double counting, because applied mitigation measures will

Table 2 presents the data sources used to derive the average values for the different categories. The full references and a summary of the results of the different studies can be found in the literature review (Volume I of this report).

Table 2: Sources for the Economic Data on Erosion

Cost category		Set	Authors
On-site costs	PC	Upper Bound (Unadjusted)	Crosson 1997, den Biggelaar et al. 2001, Eastwood et al. 2000, Evans 1996, Hartridge and Pearce 2001, Hopkins et al 2001, Mallawaarachchi 1993, Riksen & De Graaff 2001, Science Council of Canada 1986, Xu & Prato 1995 French and UK case studies on erosion (Volume II of this report)
		Mean (Adjusted)	Crosson 1997, den Biggelaar et al. 2001, Eastwood et al. 2000, Evans 1996, Hartridge and Pearce 2001, Mallawaarachchi 1993, Xu and Prato 1995 French and UK case studies on erosion (Volume II of this report)
		Lower Bound	Crosson 1997, den Biggelaar et al. 2001, Xu and Prato 1995
	MC	Upper Bound (Unadjusted)	Alcock 1980, King and Sinden 1988, Niskanen 1998, Ehrnsberger 2000
		Mean (Adjusted)	Alcock 1980, King and Sinden 1988
		Lower Bound	None
Off-site costs	SC	Unadjusted (see procedure below)	Fox and Dickins 1988, Mallawaarachchi 1993, Evans 2004, Ehrnsberger 2000, Pretty et al. 2000, Clarke et al. 1985, Eastwood et al. 2000, Evans 1996 UK Case study on erosion (Volume II of this report)
	DC	Upper Bound / Mean	Eastwood et al. 2000, ICONA 1991 French case study on erosion (Volume II of this report)
		Lower bound	None

4.1.2.3 Assessment of the Private, On-Site Costs of Erosion (PC)

The following Table 3 summarises the estimations for the **private, on-site costs of erosion** derived from the literature review and the case studies carried out as part of this project (Volume I and Volume II of this report). The studies that were excluded in moving from the unadjusted average to the adjusted average are marked in italics.

Table 3: Estimates of the Private Costs of Erosion (PC), in €$_{2003}$

Author	Year	Region	Cost per	Lower bound	Mean	Upper Bound	Comments
Darmendrail et al.	2004	England / Wales	ha*y		2.59 €		lost outputs and inputs
Darmendrail et al.	2004	Pays de Caux / F	ha*y	8.00 €	16.00 €	90.50 €	
Darmendrail et al.	2004	Lauragais / F	ha*y		36.00 €		
Hartridge and Pearce	2001	England / Wales	ha (NPV)		8.36 €		Nationwide average

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reduce the suffered impacts. The same is true for social costs (SC) and defensive measures (DC). To avoid this, it would be necessary to obtain the actual expenditure for mitigation measures rather than extrapolated figures, as well as information of how mitigation measures reduce the suffered damage.

Author	Year	Region	Cost per	Lower bound	Mean	Upper Bound	Comments
Evans	1996	England / Wales	ha*y		0.55 €		lost output
Riksen & De Graaff	*2001*	*UK*	*ha*y*		*38.48 €*		*with conservation measures*
					63.75 €		*no conservation measures*
Xu and Prato	1995	US	ha*y		0.17 € 0.26 € 0.35 € 0.42 €		at erosion rate of 2 t/ha/y at erosion rate of 3 t/ha/y at erosion rate of 4 t/ha/y at erosion rate of 5 t/ha/y
den Biggelaar et al.	2001	US	ha*y		0.47 €		Nationwide average
Crosson	1997	US	ha*y		0.69 €		Nationwide average
Eastwood et al	2000	New Zealand	ha*y		1.37 €		farm infrastructure damage
Hopkins et al.	*2001*	*US*	*ha*y*	*0.01 €*	*1.68 €*	*3.35 €*	
Mallawaarachchi	1993	NSW / Australia	ha		6.01 €		
Eastwood et al	2000	New Zealand	ha*y		6.89 €		lost output
SCoC	*1986*	*Canada*	*ha*y*	*8.23 €*	*9.60 €*	*10.98 €*	

In addition, the conservative estimate for the on-site impacts of erosion was calculated as the arithmetic average of the numbers provided by Crosson 1997, den Biggelaar et al. 2001 and Xu and Prato 1997, as these where the lowest numbers in the set. The situation described in these North American estimates represents a conservative lower-bound estimate for the situation in Europe. This gives the following values for the private, on-site costs of erosion:

Table 4: Private Costs of Erosion (PC) (Cost as €$_{2003}$ / ha*y)

Estimate	Value
Upper-bound estimate (unadjusted mean)	11.06 €
Intermediate estimate (adjusted mean)	7.56 €
Lower bound estimate	0.51 €

4.1.2.4 Assessment of the On-Site Mitigation Costs (MC)

The assessment of the On-Site mitigation costs (MC) to counter the effects of soil erosion is based on the four estimates presented below.

Table 5: Estimates of the On-Site Mitigation Costs of Erosion (MC), €$_{2003}$

Author	Year	Region	Cost per	Lower bound	Mean	Upper Bound	Comments
Alcock	1980	Queensland, Australia	ha*y		2.18 €		
King and Sinden	1988	NSW / Australia	ha*y		3.45 €		
Ehrnsberger	2000	Bavaria, Germany	ha*y		82.00 €		related to 8 t eroded soil/ha*y

The table above shows that the **data situation for mitigation costs (MC) is unsatisfactory**. Only one of the three studies listed is European; only one is fairly recent; and there is a substantial discrepancy between the results of the European and the non-European studies. Due to these concerns, and due to the methodological difficulty of extrapolating MC and PC jointly, the lower-bound estimate for the mitigation costs was set equal to zero (see also footnote 9). In moving from the unadjusted upper-bound set to the adjusted intermediate set, the Ehrnsberger study was excluded as it exceeded the results of the other studies by far.

Table 6: Mitigation Costs of Erosion (MC) (Cost as $€_{2003}$ / ha*y)

Estimate	Value
Upper-bound estimate (unadjusted mean)	29.24 €
Intermediate estimate (adjusted mean)	2.86 €
Lower bound estimate	0.00 €

4.1.2.5 Assessment of the Social, Off-Site Costs of Erosion (SC)

For assessing the social costs of erosion a slightly different procedure was used. The category of social costs is in itself heterogeneous, as it comprises different sorts of impacts. From the literature, average values were derived for eight types of impacts:

- Cost of sediment removal;
- Infrastructure damage (roads and water supply);
- Water treatment;
- Property damage;
- Flood damage;
- Impact on recreational functions;
- Climate change impacts of organic matter (OM) loss; as well as
- Economic second-order effects of erosion-induced income losses.

For each of these, between one and five estimates were available from the literature. Table 7 provides an overview of the average values (cost per ha and year) for each category. The sum of these eight average values was used as the upper-bound estimate. A conservative, adjusted mean estimate was calculated by eliminating outlier values from the set and by eliminating those values that were based on only one observation. Some of the eliminated values are substantial; therefore the adjusted average value only amounts to about half of the unadjusted, upper-bound value.[10] Finally, the lower-bound estimate was based on the lowest values for each of the categories included in the adjusted set. These values were summed up to yield a lower-bound estimate of € 21.43 / ha*y.

[10] It should be noted that there is only one quantified reference for climate change impacts Pretty et al. (2000). This underlines the uncertainties involved, and also explains much of the difference between the mean value and the adjusted mean value. The climate change impact of lost organic matter accounts for more than a third of the unadjusted estimate. However, while Pretty et al. mention that erosion can be a cause of declining organic matter content in soils, the contribution of erosion to organic matter decline is not quantified in their article. Therefore it is difficult to estimate which share of the € 60.12 / ha they report is actually attributable to erosion.

Table 7: Social Costs of Soil Erosion (€$_{2003}$ / ha, average values)

Cost Type	References	mean	adj. mean
Cost of sediment removal	Clark et al. 1985, Darmendrail et al. 2004, Eastwood et al. 2000, Ehrnsberger 2000, Fox & Dickson 1988	32.25 €	11.45 €
Infrastructure damage	Darmendrail et al. 2004, Eastwood et al. 2000, Evans 2004, Pretty et al. 2000	4.59 €	4.59 €
Water treatment	Clark et al. 1985, Darmendrail et al. 2004, Eastwood et al. 2000, Evans 1996, Pretty et al. 2000	40.26 €	40.26 €
Property damage	Eastwood 2000, Evans 2004, Evans 1996	1.35 €	1.35 €
Flood damage	Clark et al. 1985, Eastwood et al. 2000	8.49 €	8.49 €
Recreation	Clark et al. 1985, Darmendrail et al. 2004	19.78 €	19.78 €
OM loss / climate change	Pretty et al. 2000	60.12 €	
2nd order economic effects	Mallawaarachchi 1993	2.26 €	
Upper-bound (unadjusted) estimate		**169.10 €**	
Intermediate (adjusted) estimate			**85.92 €**

Table 8: Social Costs of Erosion (SC) (Cost as €$_{2003}$ / ha*y)

Estimate	Value
Upper-bound estimate (unadjusted mean)	169.10 €
Intermediate estimate (adjusted mean)	85.92 €
Lower bound estimate	21.43 €

4.1.2.6 Assessment of the Defensive Expenditure to Counter Off-Site Impacts (DC)

For the assessment of the **defensive expenditure** to counter off-site impacts, **only four estimates were available**. Eastwood et al. 2000 and ICONA 1991 report the annual public expenditure for programmes to mitigate the impacts of erosion in New Zealand and Spain. The French case study on erosion in Volume II of this report provides some evidence on the cost of protective measures installed to prevent and limit off-site impacts from erosion.

Due to the small number of estimates, an adjustment of the mean value was not possible. Instead, an average value of 25.87 € / ha*y was applied both as the upper-bound and the intermediate estimate. The lower-bound estimate was set equal to zero to take account of the limited data availability and the methodological difficulty associated with adding up extrapolated results for DC and SC (see also footnote 9).

Table 9: Defensive Expenditure caused by Erosion (DC) (Cost as €$_{2003}$ / ha*y)

Estimate	Value
Upper-bound estimate (unadjusted mean)	25.87 €
Intermediate estimate (adjusted mean)	25.87 €
Lower bound estimate	0 €

Figure 3 below gathers the information on the different cost categories in graphic form. The black dots and the Euro values in the figure indicate the intermediate estimate. The black lines above and below the dots indicate the range of the upper-bound and the lower-bound estimates, as presented above.

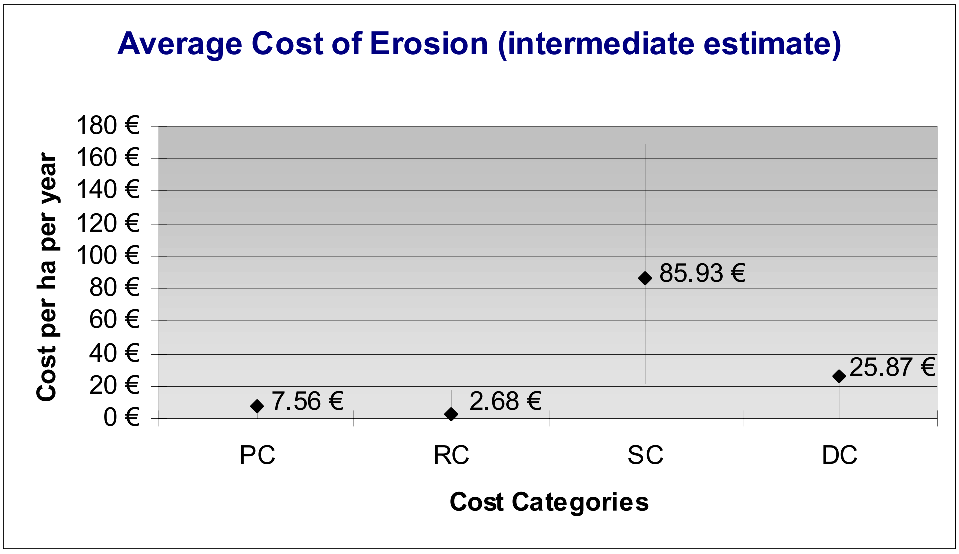

Figure 3: Average Cost of Erosion for all Categories and Estimates

4.1.2.7 Estimation of the Area Affected by Erosion

For the estimation of the area affected by erosion, the plot database assembled based on real erosion data was used (see Cerdan et al. 2003 and Volume II of this report). From this database, several land use categories were excluded – either because they were not affected by erosion, or because the total surface area in these categories was negligible. The following six categories were excluded: areas without soil cover, rice fields, orchards, post-fire areas, wetlands and forests.[11] Taken together, these six categories account for 7,106,380 t of eroded soils (per annum), which equals a 2.1 per cent of the estimated total annual erosion (337 million t). The database does not allow to estimate the impact of erosion on abandoned land (see Box 5 below), which would fall either under arable land or shrubs.

[11] To the three following categories a rate of 0 tons/ha*y was assigned: The category **No soil** comprises CORINE land cover categories like Bare rocks, Anthropogenic areas (urban areas, road and rail networks), water bodies, beaches, dunes and sands, and glaciers. The category **Rice fields** comprises the CORINE land cover category rice fields. The category **Wetlands** comprises the CORINE land cover categories inland marshes and peat bogs. An erosion rate of zero was assigned to these categories as they represent areas were there is no soil (e.g. bare rocks) or flat and waterlogged areas (e.g. rice fields). Post-fire areas were excluded because they only covered a small surface area. Consequently, despite a high erosion rate of 1.54 t / ha*y, they only accounted for about 0.1 % of total erosion. These areas may however be more relevant as “hotspots” on a local scale.

Table 10: Areas affected by Erosion

	Total area	Av. erosion rate	Total erosion	Share A*	Share B**
	ha	t / ha*y	t / y	%	%
Arable land	55,150,000	4.34	239,185,550	71.0%	69.5%
Vineyards	2,920,000	19.97	58,312,400	17.3%	16.9%
Complex cultiv. pattern	36,170,000	0.50	18,157,340	5.4%	5.3%
Grassland	32,120,000	0.29	9,282,680	2.8%	2.7%
Shrubs	24,150,000	0.50	12,123,300	3.6%	3.5%
Subtotal A	**150,510,000**	**2.24**	**337,061,270**	**100.0%**	**97.9%**
Excluded:					
No soil (bare rock etc.)	14,100	0.00	0		0.0%
Rice fields	70,000	0.00	0		0.0%
Orchards	5,180,000	0.05	269,360		0.1%
Post fire	220,000	1.54	339,020		0.1%
Wetland	1,270,000	0.00	0		0.0%
Forest	64,980,000	0.10	6,498,000		1.9%
Subtotal B	**71,734,100**	**0.10**	**7,106,380**		**2.1%**
Total A+B	**222,244,100**	**1.55**	**344,167,650**		**100.0%**

Source: Cerdan et al. 2003, Darmendrail et al. 2004.

* Share A denotes the share of the eroded load in the respective category relative to the total eroded load in the five categories included in the calculation

** Share B denotes the share of the eroded load relative to the total eroded load across all eleven categories.

Box 5: Land Abandonment and Soil Erosion

Abandonment of agricultural land in Mediterranean areas is increasing. Where it occurs, it is often the result of a number of factors:

- socio-economic factors (technological advances, globalisation of the economy, changing consumer demands, urbanisation, higher costs of living in rural areas);
- policies like the Common Agricultural Policy (CAP), and the creation of the common EU market; and
- biophysical reasons (climate change and land degradation).

The consequences of land abandonment on soils, vegetation and erosion are not clear. On the one hand, the rural depopulation and land abandonment have reactivated the natural vegetal regeneration processes and may have positive effects if fire erosion can be controlled. The establishment of semi-natural vegetation might improve soil properties and decrease runoff and erosion (Gonzales-Bernaldes 1991).

On the other hand previous soil and water conservation measures, such as terraces, are not maintained anymore and might collapse, leading to increased erosion. Feedbacks and connectivity between soil, vegetation and erosion make this even more complicated. Furthermore, these processes can act in various ways at different spatio-temporal scales.[12]

In other parts of Europe, land abandonment has been induced by political changes as in Eastern Europe. In Latvia, for example, land abandonment is regarded as one of the most pressing soil problems, which has increased up to 19% of agricultural land area in 2000. Consequences of land abandonment include the destruction of drainage systems, degradation of water ecosystems, reduction of biodiversity and decline of rural landscapes. Effects on erosion have not been observed, however (Busmanis et al. 2001).

It is therefore not possible to give a general, average erosion rate for abandoned land as it depends on the:

- Type of land use which has been abandoned (flat fields or terraces);
- Time since abandonment;
- Natural vegetal regeneration rates; and
- Climate.

It is also very difficult to have an idea of the geographical extent of the phenomena (except from local statistics for a certain study site). In the present study, land cover classes result from a reclassification of the CORINE land cover where “abandoned land” does not appear as such. The reason is that abandoned land does not correspond to one specific land cover type. It ranges from “arable land” for newly abandoned fields to “shrubs” or “forest” for fields which were abandoned a certain time ago.

4.1.2.8 Accounting for the Intensity of Erosion

The point of reference presents a problem in accounting for different intensities of erosion. The literature often leaves it unclear under which conditions results have been obtained. Many estimates are based on national averages, and only few studies identify precisely the reference conditions such as soil types and structure or crops (see also footnote 7).

This means that it is **not possible to arrive at a dose-response damage function** based on the available evidence from the literature. Instead, to apply the average results from the literature to soil uses other than agricultural land, a reference point of 4.33 t / ha*y was assumed.[13] This corresponds to the average erosion on arable land in the BRGM database.

From this reference point, the average value for erosion on agricultural land was transferred to other land use categories, accounting for different erosion intensities. For this calculation, a linear relation between the severity of erosion and its impacts was assumed.[14]

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[12] See the website of the RECONDES project, http://www.port.ac.uk/research/recondes/

[13] This assumption can be justified on the grounds that most studies have either assessed cases of erosion on agricultural lands, or provided nationwide averages, covering mainly agricultural impacts.

[14] This assumption is disputable for the private costs of erosion: In practice, depending on soil specifics and the thickness of the remaining topsoil, the impacts of erosion on agricultural yield can be either progressive or degressive – meaning that the first ton of soil eroded from a plot can have a smaller or a larger impact than the last ton of soil eroded, depending on the soil type The assumed linear relation represents a middle way. For the social costs, the assumption of a linear relation between severity and impact is more straightforward, as the cost of dredging canals or cleaning streets is roughly proportional to the volume of soil eroded – as expressed in the linear relation.

The category of arable land accounts for the bulk of erosion in Europe (72 percent of the total erosion documented in the BRGM plot database). For this category, four different intensity categories for soil erosion were distinguished (i.e. very low and low erosion, moderate, high and very high erosion). These categories were defined in such a way to make them compatible with the PESERA categorisation and the Working Group on Erosion under the Soil Thematic Strategy. The distribution of these intensity categories within the area covered by the plot database is documented in Table 11 below.

Table 11: Intensity of Erosion in the Category Arable Land

Erosion rate	defined as	Frequency*	Percent	Culm. percent
very low and low	0 - 0.5 ton / ha*y	24	32.4%	32.4%
moderate	0.5 - 1 ton / ha*y	12	16.2%	48.6%
high	1 - 5 ton / ha*y	20	27.0%	75.7%
very high	> 5 ton / ha*y	18	24.3%	100.0%

Source: BRGM plot database (see Volume II of this report, Cerdan 2003).

* Frequency: the number of entries that fall into the respective categories, out of the total of 74 entries for erosion on arable land contained in the database. It has to be noted that these entries are only roughly representative of the distribution of erosion in the covered countries.

The impacts of erosion within the different categories were weighed in the following way:

- **Very low and low rates of erosion**: 6 % of the average impacts, corresponding to an average erosion rate of 0.28 tons per ha per year;
- **Moderate rates of erosion**: 17% of the average impacts, corresponding to an average erosion rate of 0.74 tons per ha and year;
- **High rates of erosion**: 100% of the average impacts, corresponding to an average erosion rate of 4.34 tons per ha per year;
- **Very high rates of erosion**: 280% of the average impacts, corresponding to an average erosion rate of 12.14 tons per ha per year.

Combining this information with the mean values for the cost of erosion yields the following values, which were used to extrapolate the cost of erosion in the category of arable land.

Table 12: Average Cost of Erosion on Arable Land (€$_{2003}$ / ha*y)

	PC			MC			SC			DC		
erosion rate	**LB**	**Ø**	**UB**	**LB**	**Ø**	**UB**	**LB**	**Ø**	**UB**	**LB**	**Ø**	**UB**
very low and low	0,03	0,49	0,72	0,00	0,19	1,90	0,57	5,59	10,99	0,00	1,68	1,68
moderate	0,09	1,29	1,88	0,00	0,49	4,97	1,49	14,61	28,75	0,00	4,40	4,40
high	0,51	7,56	11,06	0,00	2,86	29,24	8,76	85,92	169,10	0,00	25,87	25,87
very high	1,42	21,18	30,97	0,00	8,01	81,88	24,52	240,59	473,47	0,00	72,44	72,44

It should be noted that the damage estimates derived from the economic literature applied mainly – if not exclusively – to erosion on arable land. In the course of the extrapolation, the cost estimates for arable land were also applied to other land uses by adjusting to the different erosion intensities.[15] This simplification appears justified, since more than 70 % of the observed erosion occurs on agricultural land, and 17 % on vineyards. Other categories (Grassland, Shrubs and complex cultivation pattern) account for the remaining 12 %.

The plot database covers thirteen countries (Austria, Belgium, Denmark, France, Germany, Greece, Italy, Lithuania, Netherlands, Portugal, Spain, Switzerland and the United Kingdom). This means that the new Member States are currently hardly covered. This will be amended in the future (see also chapter 4.1 in Volume II of this report for a qualitative discussion of the situation in the new Member States). Due to the structure of the database, it was not possible to break down the estimated total cost of erosion for the different countries.

4.1.3 Results

Table 13 presents some an estimate of the economic impacts of erosion. The presented results include an upper bound estimate (unadjusted mean), a mean estimate (adjusted mean) and a lower-bound estimate (the most conservative estimate for each category).

Table 13: Estimated Total Cost of Soil Erosion (million €$_{2003}$)

	PC	MC	SC	DC	Total
Arable land (55,150,000 ha)					
Upper bound	610	1,613	9,326	1,427	12,975
Intermediate (adj. mean)	417	158	4,739	1,427	6,741
Lower bound	28	0	483	0	511
Vineyards (2,920,000 ha)					
Upper bound	149	393	2,272	348	3,161
Intermediate (adj. mean)	102	38	1,154	348	1,642
Lower bound	7	0	118	0	125
Complex cultivation pattern (36,170,000 ha)					
Upper bound	46	122	707	108	984
Intermediate (adj. mean)	32	12	359	108	511
Lower bound	2	0	37	0	39
Grassland (32,20,000 ha)					
Upper bound	24	63	361	55	503
Intermediate (adj. mean)	16	6	184	55	261
Lower bound	1	0	19	0	20

[15] As an example: the adjusted mean value for the private costs (PC) was estimated at 7.56 € / ha*y for arable land (mean erosion rate of 4.34 t / ha*y). For vineyards, with a mean erosion rate of 19.97 t / ha*y, the adjusted mean cost (PC) increases proportionately to 34.80 € / ha*y.

	PC	MC	SC	DC	Total
Shrubs (24,150,000 ha)					
Upper bound	31	82	472	72	657
Intermediate (adj. mean)	21	8	240	72	341
Lower bound	1	0	25	0	26
TOTAL (for 13 countries covered, 150,510,000 ha)					
Upper bound	860	2,272	13,139	2,010	18,281
Intermediate (adj. mean)	588	222	6,676	2,010	9,496
Lower bound	40	0	680	0	720
Percentage (intermediate estimate)	**6.2%**	**2.3%**	**70.3%**	**21.2%**	**100%**

Box 6: Estimating the Non-Use Cost of Erosion

Concerning the non-use cost of erosion, the literature review (Volume I of this report) found that there is hardly any quantitative evidence. The only estimates that have quantified non-user costs of soil degradation are three Australian studies. These have elicited consumers' willingness to pay for bread produced using non-erosive agricultural practices (Dragovich 1990, 1991 and Sinden 1987). The studies found that consumers are prepared to pay between 5.2 % and 10.6 % more per loaf of bread, with a mean of 7.4 %.

Under the strong assumptions that these results are representative, and that consumers' preferences and perception of erosion as a problem are comparable between Australia and Europe, these percentages could be applied to sales of bread in Europe. For example, for the case of Germany, with sales in the bakery sector of € 13.01 billion in 2002, this would yield an estimated willingness to pay ranging from € 676 million to € 1.48 billion, with a mean of € 963 m. However, given the strong assumptions involved and the very limited data base, these numbers can only serve as a very broad indication, illustrating the potential role of non-use values. Clearly, to arrive at reliable estimates, more research into the topic is required.

4.1.4 Interpretation

For the 13 countries covered in the BRGM plot database, and for the four cost categories that could be quantified, the extrapolation yields a lower-bound estimate of € 0.7 billion per annum, an intermediate estimate of € 9.5 billion per annum, and an upper-bound estimate of € 18.3 billion per annum.

For all estimates, the social costs of erosion account for the bulk of the total costs. For the lower-bound estimate, the social costs represent 94% of the total,[16] 70% for the adjusted mean value, and 72% for the upper-bound estimate. The high share of the social costs is in line with the results of the case study on erosion that was carried out as part of this project (see Volume II of this report). To put these figures into perspective, Table 14 expresses the

[16] The high share of the social costs can be explained by the fact that two of the four cost categories (MC and DC) were not considered in the lower-bound estimate.

total cost as a percentage of the gross agricultural value added of the thirteen countries covered in the database:[17]

Table 14: Cost of Soil Erosion as a Percentage of Agricultural Gross Value Added

	PC	MC	SC	DC	Total
Upper bound	0,61%	1,61%	9,28%	1,42%	12,92%
Mean (adjusted)	0,42%	0,16%	4,72%	1,42%	6,71%
Lower bound	0,03%	0,00%	0,48%	0,00%	0,51%

Source: DG Agriculture, Agriculture in the European Union 2001, own calculations.

Table 15 presents the total cost per capita for the population of the 13 countries covered:

Table 15: Annual Per-capita Cost of Erosion

	PC	MC	SC	DC	Total
Upper bound	2.30 €	6.08 €	35.18 €	5.38 €	48.95 €
Mean (adjusted)	1.57 €	0.59 €	17.88 €	5.38 €	25.43 €
Lower bound	0.11 €	-	1.82 €	-	1.93 €

Source: Eurostat 2003, own calculations. The total population of the 13 countries covered is 373.5 million, or 80.7% of the EU-25 + Switzerland.

However, some caveats apply:

- On the one hand, **adding up private costs** (PC, on-site costs of suffered impacts**) and mitigation costs** (MC, on-site costs of mitigation or remediation measures) **involves a danger of double counting**, because applied mitigation measures will reduce the private costs. The same is true for adding up social costs (SC) and the costs of defensive measures (DC). To avoid this double counting, it would be necessary to avail of the actual (rather than the extrapolated) countrywide expenditure for mitigation measures. However, such information is only available from isolated, local studies. An alternative is to exclude mitigation costs and defensive expenditure altogether; this was done for the lower-bound estimate. If they were excluded from the intermediate (upper-bound) estimate as well, the results would be 23.5% (33.4%) lower.
- On the other hand, **several parts of the total cost of erosion were not qua**ntified. This applies above all to the **non-user cost of erosion**, for which there is virtually no data (see Box 6 above), but also to the **ecosystem services** provided by soil, which belong to the social cost (SC) category. A rough estimate of the climate change impacts of erosion is included in the upper-bound estimate of the social costs. However, due to

[17] The gross agricultural value added for the thirteen countries covered equalled € 141.5 bn in 2001 (excluding Switzerland). Data were obtained from the agricultural statistics of DG Agriculture (European Commission 2001).

methodological concerns, this figure was not included in the other lower-bound and intermediate estimates (see footnote 10 above). The only ecosystem service that is fairly well documented, and included in all estimates, is the effect of erosion on the occurrence of floods, as erosion reduced soils' capacity to absorb and retain precipitation. However, other ecosystem services, like biodiversity maintenance, could not be quantified.

Balancing those two factors leads us to conclude that **the figures presented above are in fact conservative estimates of the total cost of erosion**. Also, it has to be underlined that the "upper bound" estimate presented above only provides the upper bound for *those impacts that were quantified*. If the non-user costs and the social costs were quantified more comprehensively, **the results could increase considerably**. In this sense, the true cost of soil erosion in Europe, comprising all impacts, is probably closer to the upper bound estimate presented above, or might even exceed it.

4.2 Contamination

4.2.1 Situation

Contamination is increasingly recognised as one of the major threats for soils in Europe. Since the year 2000, there has been a growing effort to assess the extent of soil contamination at the European level, particularly for local point sources (EEA, 2000b, 2002c). Evaluations focus on the number of sites (both abandoned and operating, with identification where possible of the origin of contamination - industrial, mining or military), the typology of industrial activities, and the progress made in evaluating the situation. The management of contaminated sites is a long-term and a tiered process (primary diagnosis, main site investigation, reclamation) where remediation (the final step of the approach) involves much higher financial and time resources than site investigations (first steps).

Most experts in this field acknowledge that the data available are insufficient for assessing certain variables, such as the total surface area contaminated per class of contaminant, the percentage of population exposed to the contamination, the environmental damage caused by contaminated sites, etc. This is partly because the data collected by each Member State are not comparable, due to different evaluations of tolerable/acceptable risk levels (level of risk not to be exceeded fixed by the public authority for each target, such as human health, water resources or ecosystems).

4.2.2 Method

4.2.2.1 Distinction of impacts and cost categories

The range of activities – industrial and other – likely to generate significant soil contamination is very wide. Each activity involves several types of contaminant, each having different impacts on health, environment media, and economic actors. Assessing an aggregate cost of contamination with an acceptable level of confidence would therefore involve drawing up a typology of the situations and characterising the type of costs that exist in each situation. Based on data compiled in the EIONET database (see Volume II of this report), the typology of soil-pollution activity distinguishes 11 soil-polluting activities: (i) municipal waste disposal sites; (ii) industrial waste disposal sites; (iii) industrial and commercial sites; (iv) mining sites; (v) former military sites; (vi) oil extraction and (vii) oil spill sites; (viii) power plants; (ix) manure storage; (x) other hazardous substance sites; and (xi) other soil contamination sites (shooting ranges, etc.). However, the cost data that was derived from the literature don't permit to break down the estimated total cost of contamination over the different polluting activity types. Globally, the major costs related to soil contamination are the following:

Private costs (PC)

Soil contamination generates significant private costs for the operator of the contaminated site (private costs). Two types of PC can be distinguished:

- Cost of monitoring measures and impact assessment studies that must be carried out in order to assess the extent of contamination and the risk of further contamination of other environmental media (water, sediments, air).
- Cost of worker exposure protection: workers on an operating industrial site are protected (individual and collective protection measures) for the risks in relation with the industrial activities performed at the site. Because those protections are also efficient for the risks related to the contamination due to the industrial activities, there is no additional cost due

to contamination. The workers protection costs, estimated by an evaluation of the protection expense, are included in the global investment and production costs (cf. legal obligations of the industrial manager - Starkie; Johnson, 1975). Remediation workers have specific protection (ADEME, INRS, 2002) that are taken into consideration in the reclamation costs.

- Cost of land property depreciation: It is generally estimated using a damage function approach. Due to land use restrictions, contaminated land cannot be sold by the site owner for another type of use or only at a reduced price, thus representing a loss of economic value of the industrial asset. It is fixed by the local market price (Barde, 1975). If the type of the site uses changes, then additional remediation costs, which are paid by the public actors operating in the area, may be needed. If the use is more sensitive (e.g. residential or leisure), then the remediation costs are higher (basis of the 'fit-for-use' principle), although the final property value also gains.

Mitigation and clean-up costs (MC)

When the concentration of a polluting substance exceeds a given tolerable/acceptable risk level, thus generating an unacceptable risk for water pollution or for human health, the site operator has to implement decontamination measures (with specific objectives linked to land and environment uses – the 'fit-for-use' principle). It is quite common that public actors substitute for site operators or the liable party, because the latter either cannot be found (in the case of ancient pollution) or cannot bear the remediation costs.

Decontamination may require demolishing industrial buildings, treating rubble and/or excavating and treating the upper layer of the soil, and groundwater reclamation. Such decontamination measures may be implemented for industrial sites (see for example the French contamination case study in Volume II of this report), mining sites (decontamination of tailing ponds), oil extraction and storage sites (soils contaminated due to leakage), industrial waste dumps, etc. Remediation costs are usually defined on the basis of the 'fit-for-use' principle used in all European countries for reclaiming contaminated land. This principle, by definition, does not mean returning the site to pristine conditions (CLARINET, 2002), which is impossible to achieve in old industrial regions (such as Nord-Pas-de-Calais in France, or the Rhine Valley in Germany) where the original soils have been modified for several centuries (see Box 1).

Social costs (SC)

Soil contamination may also generate significant costs for third parties (social costs), particularly when the contamination extends beyond the area owned by the polluter (i.e. industrial with off-site effects or diffuse contamination). Although the private operator is liable for the costs generated by off-site damages, the compensations to third parties may however be paid by public actors when the private operator no longer exists as a legal entity – or cannot pay for the costs (bankruptcy). The main types of social costs are outlined below.

- The population may enter into contact with contaminated soils through different activities, such as gardening, use of playgrounds, consumption of contaminated natural products (e.g. mushrooms), etc. Depending on the type of contaminant and the level of exposure, this may result in an increase in the number of illnesses, with the associated costs of health treatment and wage loss for the concerned workers.

- Contaminated agricultural land becomes unusable for crop production or cattle breeding, leading to a loss of farm income and a loss of agricultural property value. Farmers may be compensated for the loss of income, or have their contaminated land purchased and turned into forest.

- Public authorities may impose restrictions on land use in order to reduce public exposure to contaminants (certain areas cannot be used for public or commercial constructions), which results in a loss of land property value.
- The value of real estate located in a contaminated area may decrease, as the population fears suffering health impacts. A decrease in property value must be considered as another social cost of soil contamination.
- Soil contamination by substances having a serious impact on health (radioactive nuclides, heavy metals, pesticides, etc.) may also generate significant costs related to illness. These costs can be assessed in terms of health treatment costs, loss of wages for the workers, cost for insurance companies, etc.
- Substances present in the soil may contaminate adjacent surface water bodies (through the erosion of contaminated soil) and their sediments. The negative impact on surface-water-related ecosystems (fish population, vegetation, etc.) is perceived as a cost suffered by the population (fish not suitable for human consumption due to a high concentration of hazardous substances for instance).
- Depending on the nature of the contaminant, groundwater may also be contaminated by rainwater percolating through the soils. The risk of groundwater contamination depends on the vulnerability of the aquifer. Drinking water catchments located nearby may be affected and have to shut down their wells or install water treatment units to remove the pollutant (active carbon filters to remove heavy metals and pesticides, aeration towers to remove solvents, etc.).

Defensive costs (DC)

In order to prevent any further contamination of other environmental media (water, sediments, air), defensive measures are frequently implemented. For instance, pumping wells may be drilled around the contaminated site to prevent plumes of contaminated water from expanding. Contaminated soils may also be covered (by clay layers or geomembranes) to prevent the emission of dust (wind erosion) or to hinder soil leaching by rainfall.

Non-use costs (NC)

Soil contamination can be perceived by the population as a loss of common heritage, soil being a major natural resource to be transmitted to the future generation. This loss due to non-use value can be assessed through contingent evaluation studies that aim at estimating citizens' willingness to pay to prevent soil degradation.

4.2.2.2 Accounting for the diversity of contamination types and effects

The typology of the costs per type of polluting activity described above does not capture the diversity of situations encountered in reality. Indeed, the economic impact of soil contamination is highly site specific. The costs generated by contamination depend on the following factors:

(i) The type of contaminant, which can be harmful or unharmful to human health, soluble or non-soluble in water, etc.;

(ii) The type of contamination (point source versus diffuse pollution), the extent (a few hectares to several square kilometres) and the intensity (which can be characterised using the EEA classification system);

(iii) The natural characteristics of the contaminated site: type of soil, vulnerability of groundwater resources, proximity of rivers and other surface water bodies, climate (winds directed towards cities or not), etc.;

(iv) The economic characteristics of the contaminated site and surrounding area: urban or rural area, exposed population and potential damage to properties;

(v) The duration for which the site has been contaminated, which determines the intensity of the off-site physical impacts (several years to hundreds of years);

(vi) The remediation and protection measures that have already been implemented to prevent off-site effects.

Assessing the cost of soil contamination at an aggregate level requires taking into account the diversity of situations – a real challenge, given the poor level of data available at European scale on the variables listed above. A sound methodology for assessing the cost of land contamination, taking into account the diversity of situations, was developed to serve as a framework for data collection (see annex 1). This methodology could not be applied in full, given the poor level of data availability at European level.

4.2.3 Empirical estimation

Despite the lack of data pointed out above, an attempt was made to roughly assess the total cost of soil contamination at European level. The following results have to be taken with caution as they rely on expert-based assumptions.

Extrapolation was only done for certain categories of costs, as identified in the Literature review, due to the lack of environmental data at the European level and insufficient economic references (case studies providing costs that can be used for extrapolation).

4.2.3.1 Assessment of Private Costs (PC)

There are three major types of private, on-site cost (PC) in relation with specific contamination problems. Costs of environmental survey of the operating industrial activity (in relation with the implementation of the IIPC Directive) are not included being part of the process costs.

- **Monitoring costs**. These represent the bulk of private costs when the site is still in operation (Barde 1991) (industrial environment activity reports). They largely consist of environmental impact monitoring costs particularly groundwater monitoring costs. In NICOLE experience, the costs of remedial action, when compared with risk assessment show that environmental impact monitoring brings about very significant cost savings (even an order of magnitude). This can explain why the monitoring costs are essentially borne by the industrial sector. Because a contaminated site generates a significant risk for groundwater bodies, other actors (in particular from the drinking water sector) may have to implement specific monitoring programmes. The cost of additional monitoring is then borne by third parties (included in social costs).

- **Loss of property value**. This is only shown when the site sees a change of use. (i) When a change of owner occurs, if the site remains industrial, the cost due to contamination is insignificant when compared to the process and building values. (ii) This initial investment is recovered during the exploitation period – soil contamination often results from several decades of activity (95% of the French sites identified in the BASOL database). If the site sees a change of use, the rise in property value (which depends on local land market – see Antai 2003)[18] can be balanced by the remediation costs. If the new use is more sensitive (e.g. residential or leisure), then the remediation costs are higher (basis of the 'fit-for-use' principle) but the final property value also gains. This is demonstrated e.g. by the case of the Ceramique site in Maastricht, NL, documented in Volume II of this report. There, costs of € 6.8 million were estimated for clean-up based on a function-oriented approach, as compared to a € 408.4 million for a residential redevelopment project).

In the following, only the first type of cost is assessed, based on French figures.

The first step of the extrapolation consists in assessing the number of contaminated sites where environmental and groundwater monitoring is actually implemented. In France, around 1,500 operating industrial sites have been considered as taking priority by the Ministry for Environment and are under strict monitoring (IPCC installations). A more detailed and precautionary approach has been adopted in certain regions to estimate the pressures on water resources as required by the Water Framework Directive[19]. This is the case, for instance, in the French Nord-Pas-de-Calais Region (where the MetalEurop Case Study is located) where 50 industrial sites (both abandoned and operating) have been identified as causing significant pressure on the main chalk aquifer. These 50 sites, submitted to groundwater quality monitoring, represent 1% of the total number of active industrial sites in Nord-Pas-de-Calais (all types of activities).

On the basis of the Nord-Pas-de-Calais Region, one of the most industrialised regions in France and Europe, with an important and unique groundwater resource used for drinking water supplies, and on the current situation in France where 3500 operating industrial sites are regularly surveyed (among the 68000 IIPC sites in operation), we assume that groundwater monitoring is implemented for 0.5 to 1.5% (upper-lower boundary estimates) of all industrial sites. The EEA has identified 1.5 million contaminated sites. This represents a total number between 7,500 and 22,500 sites in Europe (EEA, 2000b).

The assumption of 1.5% of industrial sites surveyed for groundwater quality issues could be considered as insufficient. To estimate the real range of industrial sites needing groundwater survey, we would need to identify the number of industrial sites potentially at risk for groundwater resources. For this, we need to know whether or not:

- Groundwater resources are located beneath the industrial sites,
- The resources are vulnerable (possibility of transfer),
- The resources are tapped (possibility of impact on targets).

There is currently no estimation of the situation at national or European level. Therefore, we have used the specific situation of an industrial region to derive the PC costs.

The second step consists in extrapolating the total monitoring cost. In the absence of more accurate data, we use, as a basis for extrapolation, the MetalEurop case study where groundwater-monitoring costs were estimated at €12,000 per year. Following an effective

[18] Antai's study is based on a survey of property value depreciation in Sweden. The analysis shows that the effects of soil contamination on property value must be based on local market data.

[19] The level of precaution taken in certain regions is higher than what is strictly imposed by the Ministry of Environment.

degradation in the quality of the groundwater used for drinking water supply, a relatively high number (12) of piezometers and wells (both on- and off-site) are used for monitoring four times a year This is compensated – in terms of cost, by a relatively low number of monitored substances (lead, cadmium, but also total hydrocarbons and some solvents) as compared to certain other sites. It therefore seems reasonable to use this case study for extrapolation. Based on these assumptions, the total estimated environmental monitoring cost ranges between **€ 96 and 289 million** per year.

Table 16: Estimates of the private costs of contamination (PC)

Estimate	Number of contaminated sites (in % of all industrial sites)	Value (million $€_{2003}$/y) (1)
Upper-boundary estimate	22,500 (1.5%)	289
Mean	15,000 (1%)	193
Lower-boundary estimate	7,500 (0.5%)	96

(1) Value reported in $€_{2003}$ using the consumer prices index (IPC) in EU25 (+7.2% between 1999 and 2002)

These costs do not include the control of groundwater quality related to the implementation of the IPPC directive. There is a risk of overlapping if the substances to be measured for the contamination monitoring are the same as those used/produced by the industrial activities. However, this is not always the situation faced by the European Countries where groundwater contamination linked to soil contamination is usually related to past activities.

4.2.3.2 Assessment of Mitigation Costs (MC)

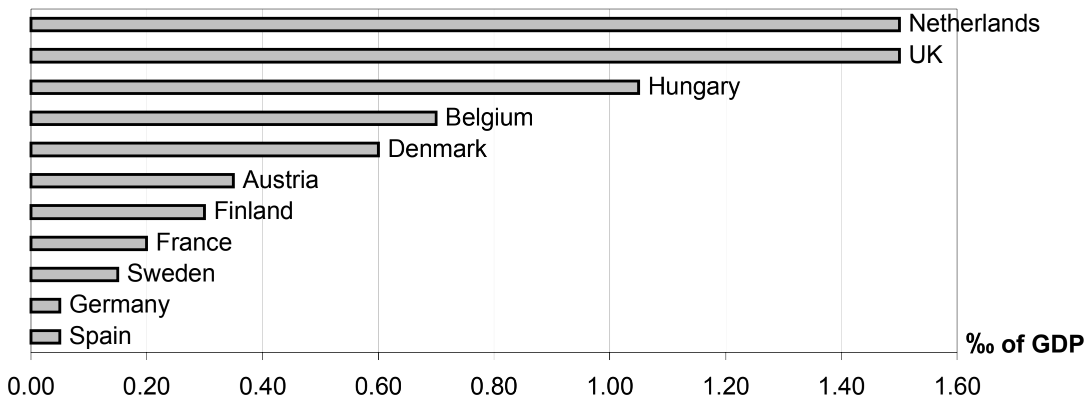
This category of costs comprises on-site costs of the reclamation and clean-up of contaminated land. Most contaminated sites are cleaned up after the closure of the industrial site if there is no risk for the workers and the environment. In cases of orphan sites or insolvency of the liable polluter, then the reclamation costs must be borne by public actors.

For the assessment of the on-site mitigation costs (MC), two slightly different approaches were implemented.

- Assessment of cost based on current expenditure

The first approach is based on published EEA statistics assessing Member States expenditure on remediation of contaminated sites in Europe. The EEA has estimated public expenditure made by 14 European countries (including 11 Member States) as a proportion of the Gross Domestic Product (GDP) in 1999. Reported values range between 0.05‰ (Spain) and 1.5‰ (Netherlands), with 8 countries spending less than 1‰. For other European countries, for which no data were available, we have assumed that expenditure for remediation is equal to the average value of **0.59‰** of GDP (average of EEA figures for EU Member States, see Figure 4 below). The annual cost is computed for each Member State and **the total cost is estimated at € 3,400 million**.

Figure 4: Expenditure on remediation of contaminated sites as ‰ of GDP_{1999}[20]



- Assessment of costs based on annual remediation expenditure

The previous approach, however, suffers from a major caveat: the annual expenditure of each Member State for remediation is indeed not proportional to the financial resources that would actually be needed to remediate all contaminated sites. On the contrary, this is rather determined by political considerations.

As a result, the time needed for the total remediation of contaminated sites may significantly differ from one country to another, depending on the priority attached to this issue and the related financial effort made by the government.

This is clearly illustrated by the EEA figures, which assess the total remediation costs and compare these to the actual annual expenditure (Figure 5). The results show that average annual expenditure corresponds to 2.5% of the estimated total remediation costs (average over 7 countries); with a maximum of 7.9% in Denmark and a minimum of 0.6% in Sweden. Put differently, these figures suggest that complete remediation will be carried out over a 12- to 13-years period in Denmark (7.9% during 12.7 years = 100%) while the same result will only be achieved over 166 years in Sweden (0.6% during 166 years = 100%). This difference is not only due to the current political desire to solve the problem, but also to the extent of contamination – which is a heritage of the past. Current accidents leading to contamination are covered by the IPPC Directive and must be treated immediately. Related costs are taken into consideration in the operating costs.

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[20] Source : EEA 2003b. Note that the figures used here are taken from the EEA web site: the indicator fact sheet "Expenditures on remediation of contaminated sites" - Fig. 2. The same figure published in the EEA report "Europe's environment: the third assessment" (2003) was not correct. The order of the bars of the graph was inverted and the data are expressed in ‰ instead % of GDP. The EEA is informed and is in the process of preparing a corrigendum.

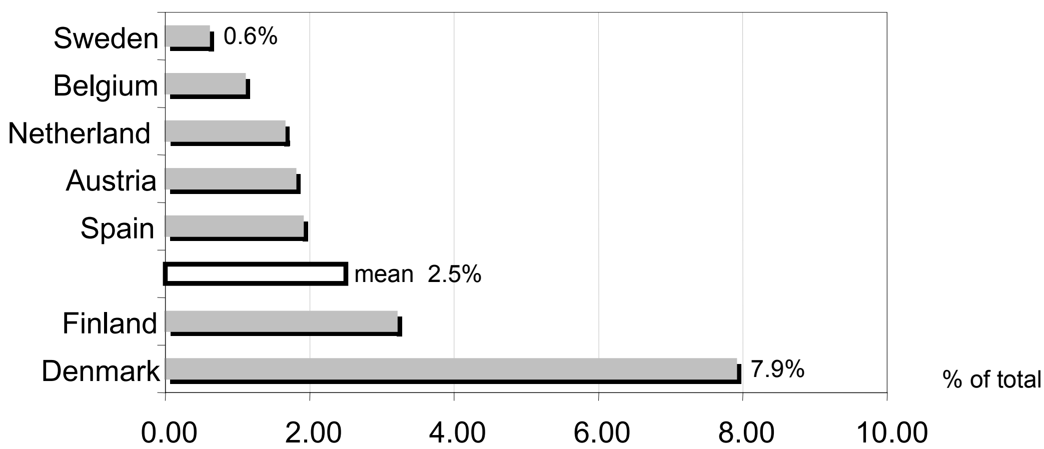

Figure 5: Expenditure on contaminated sites compared to the estimated total remediation cost in selected countries[21]

Using this as a starting point, we calculated the average annual cost that each Member State would have to bear if the entire decontamination process was to be carried out over a fixed period. Three assumptions were made concerning the duration of remediation (15, 30 and 50 years) and used to assess lower, intermediate, and higher boundary cost values.[22] Fifteen years corresponds roughly to the time needed to remediate all sites for countries having made real financial efforts in that direction (e.g. Denmark), whereas 50 years corresponds to the case of Spain, Austria and the Netherlands (see Figure 5 above).

This average annual cost is estimated as follows:

$$RC_i^{annual} = \frac{\alpha_i * GDP_i}{\beta_i * T}$$

Where: i is the index of the country.

α_i is the level of public expenditure made by Member State "i" (value provided by EEA for 11 countries, average values or estimated values - Figure 4).

GDP_i is the Gross Domestic Product of Member State i.

β_i is the estimated ratio "actual expenditure / required expenditure", which gives an indication of the level of effort of the Member State to reclaim contaminated sites (value provided by EEA for 7 countries, average value for others - Figure 5).

T is the time horizon of the remediation (15, 30 or 50 years).

The results are as follows:

- For the EU-25, the **estimated total remediation costs range between € 109 and 619 billion** (Table 17).
- The **average annual remediation cost** is then calculated for each country. The national values are then added up to provide an estimate, at the European level, ranging between **€ 2 billion per year** (lower estimate assuming β=7.9% and T=50 years) **and € 41 billion**

[21] 1999 data, Source: EEA 2002b.

[22] The choice of duration could result from new legislative constraints. The case of the Water Framework Directive, which imposes that good status be achieved 15 years after its promulgation, illustrates this point.

per year (upper estimate assuming β=0.6% and T=15 years**)**, with an **intermediate estimate at € 6.7 billion per year** (Table 18).

Table 17: Estimated remediation and clean-up costs for EU Member States

Countries	GDP 1999 (million €)	Current expenditure (‰ GDP$_{1999}$)	Current annual clean-up cost (million €$_{1999}$)	β Current Exp. 1999 / estimated total cost (in %)	Estimated total remediation costs per country (million €$_{2003}$) (1)		
					assuming β= 2,5%	assuming β= 0,6%	assuming β= 7,9%
Belgium	216,137	0.75	162	1,10	15,974	15,974	15,974
Czech republic	49,683	0.59	29	β	1,292	5,384	409
Denmark	149,169	0,60	90	7.90	1,243	1,243	1,243
Germany	1,863,458	0.05	93	β	3,944	16,435	1,248
Estonia	4,349	0.59	3	β	118	493	37
Greece	107,103	0.59	63	β	2,896	12,066	916
Spain	495,627	0.05	25	1.90	1,483	1,483	1,483
France	1,241,129	0.20	248	β	10,723	44,681	3,393
Ireland	70,719	0.59	42	β	1,989	8,286	629
Italy	1,029,991	0.59	608	β	26,906	112,110	8,515
Cyprus	7,520	0.59	4	β	203	845	64
Latvia	5,410	0.59	3	β	141	587	45
Lithuania	8,681	0.59	5	β	208	868	66
Luxembourg	15,417	0.59	9	β	405	1,686	128
Hungary	40,352	1.05	42	β	2,239	9,330	709
Malta	4,321	0.59	3	β	113	470	36
Netherlands	332,654	1.50	499	1.50	37,982	37,982	37,982
Austria	181,645	0.35	64	1.80	3,797	3,797	3,797
Poland	135,708	0.59	80	β	3,812	15,882	1,206
Portugal	93,901	0.59	55	β	2,547	10,613	806
Slovenia	17,184	0.59	10	β	545	2,270	172
Slovakia	18,699	0.59	11	β	596	2,482	189
Finland	108,215	0.30	32	3.20	1,108	1,108	1,108
Sweden	218,263	0.15	33	0.60	5,926	5,926	5,926
UK	1,170,206	1.50	1,755	β	73,562	306,509	23,279
EU 25	**7,584,164**	**0.59**	**3,969**	**-**	**199,752**	**618,510**	**109,361**

(1) The 1999 Euro values are discounted to 2003 using the ECB's annual inflation figures for each country. Assumed β values of 0.6, 2.5 and 7.9 were applied to those countries for which no evidence was available. For others, the measured data applies.

Table 18: Estimated average remediation cost (million €$_{2003}$/year) depending on the duration of the remediation programme and the assumed value of β

Estimated annual cost (million €$_{2003}$/year) with the remediation programme scheduled over T years	β = 0.6% for countries with missing data	β = 2.5% for countries with missing data	β = 7.9% for countries with missing data
Estimated total costs	618,510	199,752	109,361
T = 15 years	**41,234** [(a)]	13,317	7,291
T = 30 years	20,617	**6,658** [(b)]	3,645
T = 50 years	12,370	3,995	**2,187** [(c)]

(a) Upper-bound estimate; (b) intermediate estimate; (c) lower-bound estimate. Calculation is based on the estimated total remediation costs for EU 25.

Note that a limitation of the approach implemented above is that it only assesses the cost of the minimum clean-up actions needed for a new, different use of the site ('fit-for-use' principle), and not the cost of a full restoration to pristine conditions. The costs may differ significantly, as shown by the Dutch example below (box 1).

Box 7: The costs of 'fit-for-use' remediation and total contamination removal

In the Netherlands, 232 ha were reclaimed in 2003 for a global cost of €252 million. This figure can be compared to the hypothetical cost of total contamination removal, which could be achieved through the complete excavation of contaminated soil, transport, land-filling and temporary safety measures (during the works). The cost of this hypothetical scenario is assessed assuming (i) excavation of a 2-m layer (over 232 ha) at a cost of 50 €/m^3; (ii) a transport cost of 0.5 €/ton/km, with a range between 10 and 150 km, and (iii) land-filling costs: ranging between 60 € / ton and 450 € / ton (depending on the landfill class).

This leads to an **estimated cost of restoration to** total contamination removal that ranges between €0.60 billion (lower-boundary value) and € 4.9 billion (upper-boundary value), with an intermediate estimate at € 2.7 billion, which is **approximately 10 times greater than what was actually spent** on remediation of the Dutch sites using the 'fit-for-use' principle.

4.2.3.3 Assessment of off-site Social Costs (SC)

As explained above, the off-site social costs of a contaminated site are highly dependent on its characteristics in terms of natural, demographic and economic environment. The absence of systematic data on the characteristics of the contaminated sites, therefore, does not allow us to implement the methodological framework described in the previous sections. This in turn means that it is not possible to assess separately the cost of human health impact, environmental impact, depreciation of land value and real estates located in the vicinity of the sites, etc (see Annex 2).

A pragmatic extrapolation approach was therefore implemented to overcome this difficulty. It consists in extrapolating the results of the MetalEurop case study. As previously indicated, social costs are mainly related to off-site effects that generate costs for third parties. Off-site effects are not differentiated for most contaminated sites (e.g. less than 1% of the sites identified in the French BASOL database). This situation is more common for Megasites. However, all types of social costs are not assessed on such sites. MetalEurop is one of the rare cases for which such figures are available.

In this case study, the ratio between on-site costs (PC and MC) and off-site social costs is equal to 1/5 (French contamination case study in Volume II of this report). Given that off-site social costs are probably higher in the case of MetalEurop (a Megasite with major off-site impacts and long-term effects due to the duration of contaminant emission) than in most other contaminated sites (Megasites representing only a small percent of the total number of contaminated sites in Europe), the factor 5 is taken as the upper boundary. The lower boundary of social costs is assumed to be at least equal to the on-site costs (factor 1). Factor 2.5 is taken as a median value.[23] The aggregate off-site social costs of contaminated sites at the European level is therefore estimated (Table 19) and leads to total **social costs ranging between € 2 and 207 billion per year with an intermediate value of € 17 billion per year.**

Table 19: Total estimated social costs (SC) as a function of PC + MC (million $€_{2003}$/year)

Annual social costs	Factor 1	Factor 2.5	Factor 5
Lower bound estimate	**2,283** [(a)]	5,708	11,416
Median estimate	6,850	**17,126** [(b)]	34,252
Upper bound estimate	41,523	103,807	**207,615** [(c)]

(a) Lower-boundary estimate; (b) intermediate estimate; (c) upper-boundary estimate.

4.2.3.4 Assessment of Defensive Costs (DC)

The defensive costs are assessed as follows. We assume, as above, that only 0.5 to 1.5% of the 1.5 million contaminated sites represent a potential threat for groundwater (See 4.2.3.1). We further assume that defensive measures have been implemented, on average, at 20% of these sites where groundwater is contaminated as it is in France (Basol database). We then use the cost estimate found in the MetalEurop case study (€ 300,000 per year) to extrapolate at the European level. The total estimated cost ranges between € 450 and 1,350 million.

4.2.4 Interpretation

Contamination by industrial and other activities, considered as a major threat to European soils, is generating at the EU level a total cost that is roughly estimated at € 25 billion. Estimated social costs (SC) represent about 69% of this total cost, whereas remediation costs (MC) based on the 'fit-for-use' principle adopted by most European countries represent another 27%. PC and DC represent approximately 4% of the total cost (Table 20 and Figure 6). A large part of the remediation costs are, however, covered by the public budget – thus, from an economic point of view, they can be considered, to some extent, as social costs since they are paid by the taxpayers and not by the polluters. The upper and lower boundary cost values should be used to assess the order of magnitude of the impacts of soil contamination at European level. As mentioned above, a more detailed estimation should differentiate between the type and level of contamination (see Annex 2 for more details).

[23] The social costs for other sites in Europe may be less or higher than the social costs for the MetalEurop site. It depends on many parameters, such as duration of contamination (more than 100 years in the case of MetalEurop), number of contaminants, surface area, health monitoring, etc.

Table 20: Estimated total costs of soil contamination in Europe (million €$_{2003}$/year)

Cost categories	PC	MC	SC	DC	Total
Lower boundary estimate	96	2,187	2,283	482	5,049
Intermediate	192	6,658	17,126	965	24,941
Upper boundary estimate	289	41,234	207,615	1,447	250,585
Percentage of total cost (1)	0.8%	26.7%	68.7%	3.9%	100.0%

(1) Based on the intermediate values

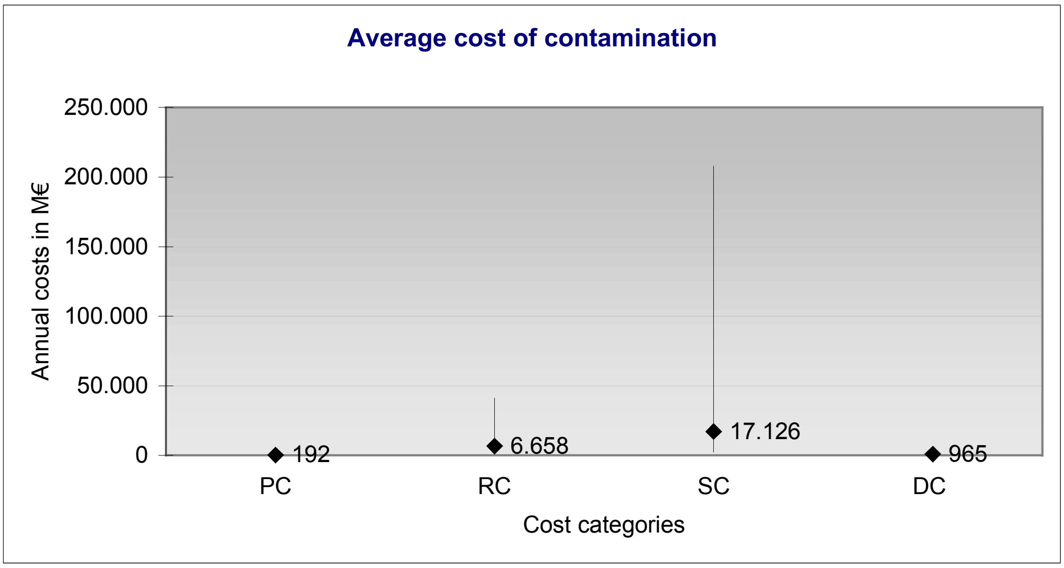

Figure 6: Total costs of contamination (million €$_{2003}$)

If remediation costs were not assessed on the basis of the 'fit-for-use' principle, but rather for a total removal of all contaminated soil (as estimated for the Dutch example in Box 7), the actual MC figures could be 10 times higher.

To put these figures in perspective, the estimated total cost is expressed per capita and as a percentage of the industrial GDP in Europe (Table 6):

- The annual total per-capita cost of soil contamination varies from 11 €/year to 552 €/year per capita with an average of 55 €/year per capita. This compares with an average per-capita GDP of 20,454 €/year in the EU-25.
- On average, the total cost of contamination represents about 0.9% of the EU25 industrial GDP, with a maximum of 9%.

Table 21: Estimated total costs per capita and as percentage of industrial GDP

Estimates	Estimated total cost (million $€_{2003}$/year)	Total cost per capita ($€_{2003}$/year) (1)	Cost as a percentage of industrial GDP (in %)(2)
Lower bound estimate	5,049	11	0.2%
Intermediate estimate	24,941	55	0.9%
Upper bound estimate	250,585	552	8.9%

(1) The EU-25 population is estimated about 454 million inhabitants (2003 - Eurostat)

(2) In 2003, the industrial GDP represented approximately 30.45 % of the EU-25 GDP (9,286 billion € - Eurostat)

Despite the lack of data pointed , these calculations present an attempt to roughly assess the total cost of soil contamination at the European level. The results have to be taken with caution as they rely on expert-based assumptions. The empirical estimation clearly reveals that economic data on contaminated sites are insufficient and that economic indicators need to be developed at national level (see also EEA 2002a).

In general, data availability on private costs (PC) category (valuation of contaminated land, cost of groundwater/drinking water treatment derived from local soil contamination, etc) is insufficient. This also occurs for the defensive costs (DC), were cost data are available only at the local or case study level. The remediation cost estimation seems more accurate than the above category since it is based on quantitative data that is available at national level. However, MC costs are closely linked to the national planning regime on environmental matters and does not reflect an estimate of the real costs. The social costs (SC) are expressed as a function of the algebraic sum of the PC and MC costs. The social costs depend on many parameters such as type of soil contamination, number of population affected, etc. In this study, the estimates of SC vary between 100% (lower-bound ratio) and 500% (upper-bound ratio) of the on-site costs. There is an expert's agreement on using these ratios as a rough estimate. Finally, the non-use costs (NC) could not be quantified in the course of this study.

4.3 Salinisation

4.3.1 Situation

Salinisation is problematic in several European countries, mostly in the Mediterranean. However, the data coverage on the distribution of salinisation is unequal. While salinisation is also a considered a problem in Italy and Greece, the European Environment Agency (EEA 2003b, p. 208) only reports data for Spain, Hungary and Bulgaria; as well as different parts of the former Soviet Union, where salinisation is a widespread problem. For European countries other than Spain, Hungary and Bulgaria, the EEA does not report any data. Consequently, the extrapolation focused on these three countries in the following.

In the non-European context, research on the economic impacts of salinisation has mainly taken place in Australia, where salinisation is perceived as a major threat to the natural environment (see e.g. Williams et al. 2002, PMSEIC 1998). Ahmad and Kutscher (1992) provide rough estimates of the economic impacts of salinisation in Pakistan.

4.3.2 Method

The extrapolation of the economic impacts of salinisation mainly considers the impacts on agricultural productivity. These impacts are calculated on the basis of agricultural gross value added per ha, using data obtained from DG Agriculture (European Commission 2001). Based on results of the Spanish case study on salinisation and the database research by BRGM, it was assumed that in cases of light salinisation, up to 10% of the output are lost, between 10 and 50% for moderate salinisation, and 50 – 90% in cases of severe salinisation. These impacts include both the effect of shifting to different crops that are more salt-tolerant, but produce a lower return, and the effect of reduced yield losses if the same crop is maintained (see Spanish case study in Volume II of this report for a detailed description). They do not, however, include the effects of salinisation on soil structure and hydraulic properties leading e.g. to higher costs of tillage.

The impacts of salinisation on agricultural output were calculated on the basis of the agricultural land area. For this, EEA data on salinisation in Spain, Hungary and Bulgaria were applied to the total agricultural area of the affected countries.[24] The information on the area affected by salinisation is presented in Table 22 below.

Table 22: Area affected by Salinisation

	Severe		Moderate		Light		Total	
	%	ha	%	ha	%	ha	%	ha
Spain	0.2%	50,850	0.8%	203,400			1,0%	254.250
Hungary	6.4%	374,651	0.8%	46,831	1,0%	58.539	8,2%	480.022
Bulgaria			0.5%	27,585			0,5%	27.585

[24] Note that the EEA data on the area affected by salinisation measures the area as a percentage of the total land area, and not only the affected agricultural area (EEA 2003b). Assuming that the effects of salinisation will mainly be felt on agricultural areas, the EEA data were applied to agricultural areas only. This means that the results will understate the total impacts, as the impacts of salinisation on non-agricultural land have been left out of the analysis.

While the Spanish case study provided information on the on-site impacts of salinisation (mainly reduced agricultural output), the off-site damage caused by soil salinisation is not described in the case study. The literature survey carried out as part of this project identified only one Australian study that had assessed these factors. This study conducted for the Prime Minister's Science, Engineering and Innovation Council considered the impacts on transport infrastructure (roads and bridges) caused by shallow saline groundwater, as well as damage to the water supply infrastructure (PMSEIC 1998). The cost reported for these impacts amounted to $€_{2003}$ 24 / ha. In the absence of European estimates, the impacts on infrastructure had to be assessed on the basis of Australian estimates. No further adaptation of the Australian values to the European context was undertaken.

In the same way, the impacts of saline soils on environmental assets were inferred from Australian estimates. These include the impacts of soil salinity on native vegetation, riparian ecosystems and wetlands, as well as knock-on effects on tourism. In the Australian case, these costs are estimated at 40% of the infrastructure costs caused by soil salinisation, which corresponds to $€_{2003}$ 10 / ha; however this should be seen as a conservative estimate of the total ecosystem impacts.[25] Again, the Australian estimates were applied to the area affected by salinisation in Spain, Hungary and Bulgaria without a correction factor, assuming that the impacts differ between Australia and Europe only in scale, but not in intensity.

The assumption made here, that infrastructure impacts and impacts on native vegetation can be transferred from Australia to the European context, is highly debatable. This move was born out of necessity, since no economic assessment of the off-site cost of soil salinisation could be obtained for Europe.[26] As the quoted estimates are regarded as an underestimation of the true cost for the Australian context, they would definitely appear as a crude estimation for the European context. In this sense, the results presented have to be viewed with caution. Also, the absence of European data on this important aspect of soil degradation clearly points to a need for economic or agronomic research.

In contrast to the calculations for agricultural output losses, the figures for infrastructure and environmental damage were applied to the total area affected by salinisation in Spain, Hungary and Bulgaria, rather than the agricultural area affected only. For infrastructure and environmental damage, no distinction was made between the different categories of soil salinisation intensity, as the Australian original data did not provide for such differentiation.

Some other limitations and assumptions should also be noted.

- First, the estimation did not explicitly account for different soil types and their vulnerability to salinisation. Implicitly, the differences in soil types between European countries are however partly reflected in the agricultural productivity per country.
- Secondly, the estimation only accounted for different agricultural uses in so far as they are reflected in the agricultural productivity per country. However, it was not assessed

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[25] Indeed, the Australian Dryland Salinity Assessment refrained from quantifying the losses of environmental and biodiversity assets in monetary terms, but mentioned that these are likely run into the 10's to 100's of millions of Australian Dollars per annum. This supports the view that the estimate of 10 € / ha used here is closer to the lower bound of the real impacts (National Land & Water Resources Audit 2000). The view that these are conservative estimates was also supported by David Dent, president of ISRIC (personal communication).

[26] The absence of economic information on these points was confirmed inter alia by David Dent, president of ISRIC (personal communication).

whether countries are particularly vulnerable to the impacts of salinisation due to their agricultural structure, including the type and mixture of crops grown, or to the distribution of crop cultivation and livestock farming.

4.3.3 Results

The following tables present the nationwide cost of salinisation for Spain, Hungary and Bulgaria. The calculations assess the cost of suffered impacts only (leaving aside the cost of mitigation measures or defensive expenditures). They present examples of the private on-site costs (PC) in the form of yield losses, and social costs (SC) in the form of infrastructure damage and environmental damage. Due to the limited evidence on SC, these calculations cannot be regarded as exhaustive. All figures are given in 2003 €. For the agricultural yield losses, figures are quoted as lower-bound (LB) and upper-bound (UB) estimates.

Table 23: Cost of Salinisation for Spain (PC and SC) (million €$_{2003}$)

	Severe		Moderate		Light	Total	
	LB	UB	LB	UB	UB	LB	UB
Agricultural yield losses	23.73	42.72	18.98	94.92	0	42.71	137.64
Infrastructure damage						12.08	
Environmental damage						4.83	
Total						**59.62**	**154.55**

Table 24: Cost of Salinisation for Hungary (PC and SC) (million €$_{2003}$)

	Severe		Moderate		Light	Total	
	LB	UB	LB	UB	UB	LB	UB
Agricultural yield losses	68.45	123.21	1.71	8.56	2.14	70.16	133.91
Infrastructure damage						18.23	
Environmental damage						7.29	
Total						**95.68**	**159.43**

Table 25: Cost of Salinisation for Bulgaria (PC and SC) (million €$_{2003}$)

	Severe		Moderate		Light	Total	
	LB	UB	LB	UB	UB	LB	UB
Agricultural yield losses	0	0	1.08	5.38	0	1.08	5.38
Infrastructure damage						1.32	
Environmental damage						0.53	
Total						**2.93**	**7.23**

The above calculations estimate the cost of suffered damages (i.e. the PC and SC categories in Figure 1). In addition, it is also possible to identify the potential costs of restoring soils affected by salinisation (i.e. the MC category). For the case of Pakistan, Ahmad and Kutscher (1992) estimated the cost of improvement measures at US$ 500 / ha. Converted into current price Euro, this is approximately equivalent to $€_{1999}$ 595 / ha. The database research on salinisation that was carried out as part of this project, citing Prof. Guiseppina Crescimanno, mentions a much higher remediation cost of € 4,500 / ha (Volume II of this report). The large difference between these two estimates can be explained due to the difference in unit labour costs between Pakistan and Italy, and the different cost of intermediary inputs for the remediation measures.

Taking these two estimates as lower-bound and upper-bound estimates, it is possible to calculate the hypothetical mitigation cost for soil salinisation in the three affected European countries (i.e. the category of MC in Figure 1). In Spain, the mitigation cost for the severely salinised agricultural areas would then range from € 30 Million to € 223 Million. For Hungary, the corresponding cost range would be € 249 Million to € 1.84 Billion. Extending this to agricultural areas affected by moderate salinisation increases the cost ranges to € 151 Million to € 1.25 Billion for Spain, and to € 286 Million to € 2.36 Billion for Hungary.

Table 26: Mitigation Cost of Salinisation (in million $€_{2003}$)

	severe salinisation		severe and moderate salinisation	
	LB	**UB**	**LB**	**UB**
Spain	30.30	249.65	151.49	1,248.24
Hungary	223.23	1,839.35	286.01	2,356.67
Bulgaria	-	-	16.44	135.43

As the affected areas are in Europe, it could be argued that the upper-bound estimates are more realistic. Also, the calculations above only include the agricultural land that is affected by salinisation, whereas non-agricultural land affected by salinisation is not covered. At the same time, it has to be underlined that the reported cost ranges are hypothetical. They are calculated on the assumption that all affected areas are desalinised through salt-leaching and improved irrigation techniques. However, in reality, this is not likely to occur:

- In many cases a complete restoration of salinised agricultural areas may not be possible, or will not be economically feasible.
- If funding for desalinisation is not available, part of the area affected by severe salinisation will probably be abandoned, or has been abandoned already, leading to a total loss of the agricultural yield and possibly to wider social impacts. These impacts would need to be investigated on the local level, but could not be assessed in this study.
- It is also likely that affected land will only be desalinised to the point where agricultural activity becomes possible, but will not be restored completely.

Finally, it has to be noted that the hypothetical mitigation cost has been extrapolated to all affected areas. Therefore, these figures cannot be combined with the costs of suffered damage (PC and SC categories) calculated above, as this would necessarily lead to double-counting. As was argued in the literature review (Volume I of this report), the costs of suffered damage could be combined with the actual expenditure on desalinisation, rather than hypothetical investments necessary to remediate soils. However, data on actual expenditure was not available.

4.3.4 Interpretation

When compared with Australian results for crop yield losses due to soil salinisation, it appears that the estimates presented above lie at the upper end of the range. However, it is also important to note that the Australian comparison values are average values, which are not differentiated for intensity of salinisation.

From the estimation of the suffered damages – both private (PC) and social (SC) – it becomes clear that the bulk of the costs is caused by yield losses in agriculture (PC). The environmental costs (SC) amount to 3 – 8% of the total, and infrastructure damage (SC) for another 8 – 20%. However, it should be kept in mind that the environmental costs were derived only on the basis of the Australian results, which are considered as very conservative estimates for the Australian context.[27]

In order to put the calculated cost estimates into perspective, Table 27 reports them as percentages of the agricultural Gross Value Added for the three affected countries.

Table 27: Cost of Salinisation as a Percentage of Agricultural Gross Value Added

	Spain		Hungary		Bulgaria	
	LB	UB	LB	UB	LB	UB
Agricultural yield losses	0.18%	0.58%	3.28%	6.26%	0.05%	0.25%
Infrastructure damage	0.05%		0.82%		0.06%	
Environmental damage	0.02%		0.33%		0.02%	
Total	**0.25%**	**0.65%**	**4.43%**	**7.41%**	**0.13%**	**0.33%**

It has to be underlined that these figures only cover two of the 25 EU Member States, as well as the accession candidate Bulgaria. This limitation is due to the fact that EEA data on salinisation is only available for the three countries in question. It does not imply that salinisation is not problematic elsewhere in Europe; in fact the estimates would be considerably higher if corresponding figures were available for e.g. Italy and Greece.

Due to their wide spread and because of their hypothetical character, the mitigation costs are less informative of the costs imposed by salinisation. They do provide evidence of the fact that the remediation of salinised soils can be costly in the short term. At up to 4,500 €/ha, the restoration cost are five to six times higher than the upper bound for the calculated yield losses (800 €/ha for severe salinisation in Spain). However, the costs of repairing / mitigating salinisation only occur once, while the yield losses occur annually. This means that it would take about six years for the costs to pay off, depending on the discount rate applied. However, this rough calculation does not consider the off-site impacts of salinisation. If these were factored in, including a more complete valuation of environmental and infrastructure damage, then the pay-off time would be even shorter.

[27] Personal communication with David Dent, ISRIC

5 Conclusions and Interpretation of the Results

The current study has assessed the costs of current soil degradation in the European Union. In doing this, the authors have looked at the eight different types of soil degradation identified in the Commission Communication "Towards a Thematic Strategy for Soil Degradation" (European Commission 2002). The analysis has assessed not only the on-site costs of soil degradation, which have traditionally been the focus of economic and especially agronomic research, but has also considered the off-site costs associated with soil degradation. The different cost categories are represented in the Figure 7 below (see also chapter 2.1 above for a detailed description).

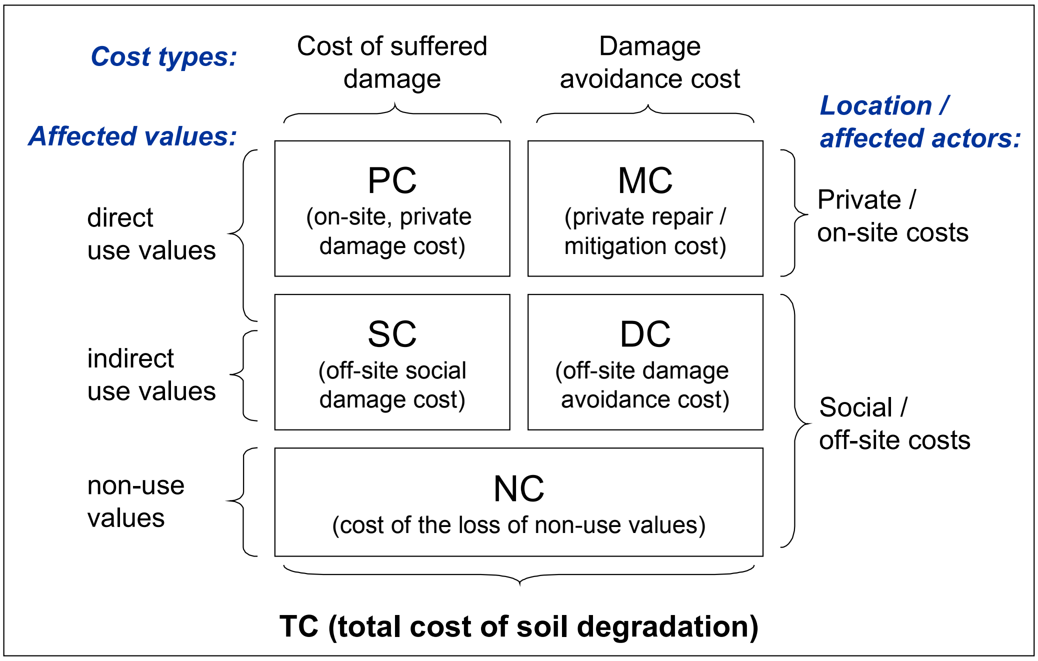

Figure 7: Overview of the different cost categories

In the figure above, the two categories at the top (PC and MC) represent the on-site costs, whereas the three categories at the bottom together constitute the off-site costs. Off-site costs are included both as the cost of suffered damages (social costs or SC in the figure above), or the cost of measures to avoid, limit or mitigate the off-site impacts (defensive expenditure, DC). The non-user costs (NC) also form part of the off-site costs, but could not be quantified in this study. In an economic sense, all off-site costs are examples of externalities.[28]

In the following, the main findings of the empirical estimation will be summarised. In doing this, we will first turn to the methodological findings and the findings on data availability, indicating where further research is needed to assess the economic impact of soil degradation in greater detail. Subsequently, we will summarise some of the quantitative results that can be inferred from this study and discuss these, also with a view to the intertemporal valuation.

[28] An externality can be defined as "the result of an activity that causes incidental benefits or damages to others with no corresponding compensation provided to or paid by those who generate the externality" (Baumol et al., 1988). In the presence of externalities, markets normally fail to deliver optimal outcomes, justifying a government intervention.

5.1 Methodological Findings

The cost categorisation that was used in this document was derived from a review of the literature, and formed the basic methodology that has guided the analysis. At the same time, it became clear from the extrapolation and the supporting case studies that the cost categories are not always clear-cut. By applying the methodology to the different types of soil degradation, the following insights were gained:

- The **category of private costs (PC)** is not applicable to contamination in the same way as to erosion or salinisation. In most cases, the immediate damage suffered by the originator of the contamination will be limited to a loss in commercial value of the contaminated site, which is a minor share of the true costs. Instead, damage accrues mainly to third parties, i.e. neighbours or downstream water users. For the polluter, costs will mostly arise if he can be made to pay for the remediation of the contamination, depending on the existing liability rules and their enforcement. Therefore, for the case of contamination, the on-site costs will mostly fall into the category of mitigation costs (MC).
- For the **category of repair / mitigation costs (MC)**, it has to be underlined that the figures reported do not represent the costs of returning soils to a pristine condition, which is impossible in most instances of soil degradation. Rather, they are an estimate of the costs that land users incur to mitigate and limit the impacts of soil degradation, e.g. through soil stabilisation measures in the case of erosion, or by returning the soil to a fit-for-use state in the case of contamination. Where MC estimates are used because estimates of the true impact are not available, they have to be regarded as a lower-bound figure. The treatment of mitigation costs differs for the various soil threats due to the availability of data: for the case of contamination, the actual expenditure on the remediation of contaminated land is documented fairly well. For salinisation and erosion, such figures are not available, the calculations presented for these threats are therefore based on the extrapolation of local estimates.
- For the **distinction between on-site costs and off-site costs**, the present study has mainly used a spatial criterion to distinguish between the two, based on where the damage occurs. However, in some instances it may also be helpful to apply an economic criterion, based on who pays for the damage. The two criteria will deliver different results in cases where on-site remediation or mitigation measures are funded from the public budget (this is often the case with contaminated “orphan sites” where the original polluter cannot be held liable). Based on the spatial criterion, the cost of these measures would be regarded as on-site mitigation costs (MC). Economically, they would be classified as off-site, social costs (SC), as they are not covered by the polluter (see also the French erosion case study in Volume II of this report). In most cases, however, the two criteria will yield the same result, as the damage that occurs outside a site is hardly ever compensated, and thus has to be borne by the public.
- The **intertemporal valuation** of soil degradation is difficult to reflect in an economic valuation. In many cases, soil degradation is essentially irreversible, so that damage would persist in eternity and become unlimited. However, because future costs are normally discounted in economic assessments, irreversibility typically escapes an economic valuation. Although this is often perceived as a shortcoming, there is no generally accepted alternative. In this document, the focus was therefore placed on the annual, current cost of soil degradation. The question of intertemporal valuation is discussed in chapter 5.5 below.

The empirical economic assessment of soil degradation attempted in this study revealed a substantial lack of data and empirical evidence in many areas (see also 5.3). To react to the lack of primary data, the methodology developed in the literature review (Volume I of this report) has been adjusted on several occasions:

- In the case of **salinisation**, not all of the data on the economic impacts was obtained as annual cost per hectare. Instead, part of the impacts was calculated based on aggregated agricultural productivity.
- In the case of **contamination**, it was not feasible to base the estimation on per-hectare figures due to the large differences between different sites, and different types of contamination (see also Annex 1). Instead, estimations were based on the number of contaminated sites and on the remediation expenditure in different European countries.
- In the case of **organic matter loss**, while sporadic evidence on the economic impacts exists, an extrapolation of the impacts was not possible due to the lack of Europe-wide data on the extent of organic matter loss.
- For all types of soil degradation, the category of **non-use values** proved to be most deficient in terms of empirical data. While it is acknowledged in theory that non-use values may represent a significant share of the total economic value of an environmental good, there is hardly any empirical data on non-use values attached to soil (see also Box 6). Non-use values were therefore not quantified for either type of soil degradation.

From a methodological perspective, the approach to assessing economic impacts of soil degradation as described in the literature review is thus valid and useful. However, it has to be applied with some flexibility: not all cost categories are equally applicable for all types of soil degradation, and not all categories can be assessed with the same degree of certainty.

In this context, soil sealing and organic matter loss through peat mining represent special cases, as they occur intentionally, and as they are inextricably linked to economic activities. The economic impacts of soil sealing and peat mining would therefore need to be considered in connection with the economic benefits derived from construction and peat mining. This could take the form of a cost-benefit-analysis or an environmental impact assessment, including an assessment of alternative options (e.g. use of brownfield sites for construction, or use of alternative sources of heat and fertiliser instead of peat).

However, this study was not intended as a cost-benefit-analysis of soil degradation. Therefore the benefits of soil-degrading activities were not considered. Especially in the cases of soil sealing and peat mining, this leaves scope for further research and assessment.

5.2 Findings on Data Availability

While the methodology is sufficiently developed in principle, substantial further research is needed to gather the necessary empirical data. In the last years, substantial efforts were made to collect and improve soil data, e.g. through the Working Groups established under the Thematic Strategy for Soil Protection, but also by the EEA and through European research projects. However, these efforts have mainly considered physical aspects of soil degradation, whereas socio-economic aspects were not a core interest.

Consequently, the quantification of different cost categories remains difficult and fraught with uncertainties. Substantial gaps exists both for physical soil data and economic data, as documented in the literature review conducted as part of this project (Volume I of this report) and the report on case studies and database research (Volume II of this report). In addition, the work of the Working Groups established under the soil thematic strategy documented the knowledge gaps that are apparent for some types of soil degradation, and for many types of impacts. This applies in particular to the loss of soil biodiversity, but also to soil sealing and compaction. The following table, based on table 14 of the Literature Review (Volume I of this report), gives an overview of the different cost components that were assessed in this study.

Table 28: Overview of Cost Components Estimated in the current Study

	Erosion	Contamination	Salinisation	OM loss	Biodiv. loss	Compaction	Sealing	Floods & landslides
PC	€	€	€	⊕	–	≈	–	€
MC	€	€	€	–	–	–	–	≈
SC	€	€	€	⊕	–	(⊕)	(⊕)	€
DC	€	€	–	–	–	–	–	≈
NC	–	–	–	–	–	–	–	–
Σ	€	€	€	≈	–	–	–	≈

With € = monetary assessment
⊕ = quantitative assessment
(⊕) = for compaction and sealing, indirect social costs arise through the effect on floods and landslides
≈ = qualitative assessment
– = no data available / only preliminary qualitative assessment

This overview table shows that there is still a need for significant improvements of the data base – both in terms of physical and economic data – before more substantiated conclusions can be drawn. The following points summarise main fields for future research:

- **Additional data requirements** concern not only the economic valuation of impacts. For soil biodiversity loss, soil organic matter loss and compaction in particular, there is also a need to better understand the effect of soil degradation on soil functions. To this end, indicators are needed to assess the state of the soil degradation, and to identify and quantify different impacts, including an assessment of socio-economic impacts.
- In order to illustrate what could be done and what would be needed, Annex 2: describes the **methodology and the information needs for assessing the economic impact of soil contamination** in greater detail. In order to gather the necessary primary data, several economic case studies would be needed. On the basis of this additional data, the economic impacts of soil degradation could be described in more detail, and with more confidence. Similar requirements could be formulated for the cases of erosion and salinisation, especially regarding ecosystem services and non-use values.
- In the case of **soil organic matter** loss, one facet that can be assessed with some confidence is the climate change impact of carbon losses from soil, e.g. by using cost information obtained through the European emissions trading scheme. Comparable calculations for the UK are discussed in the Annex chapter on Organic Matter loss (see A-1.1). However, a Europe-wide estimation of these effects is not possible due to the lack of coherent data on soil organic matter loss.

- For the case of **floods and landslides**, the problem arises that this threat is closely connected to other types of soil degradation. In particular, floods and landslides can be caused or aggravated by soil sealing, compaction and erosion in upstream areas, leading to reduced infiltration of rainwater and increased runoff. While this linkage is supported by hydrological research, it has not been quantified in a comprehensive manner. In particular, the impact of sealing, compaction and erosion on the costs caused by floods and landslides still needs to be investigated, e.g. through a number of case studies.

5.3 Quantitative Findings

Given the lack of empirical data that is apparent for many threats, and for many cost categories, the quantitative results presented in this study have to be interpreted with caution. Even for threats like erosion and contamination, which have been researched in greater detail in recent years, tremendous gaps still exist when it comes to placing a monetary value on the observed damage.

Despite these limitations, the current study has provided tentative estimates of the economic impacts of soil degradation in Europe, which may serve to illustrate the dimension of the problem. Table 29 presents the range of estimates calculated for the different cost categories, for three types of soil degradation. Two points should be noted:

- The numbers below should still be regarded as ***conservative estimates***, because many impacts could not be quantified at all. Hence the values reported as upper bounds in the table below do not provide the upper bound for all impacts of soil degradation, but merely the *upper bound for those aspects of soil degradation that were quantified in monetary terms in this study*. The real costs of degradation, including impacts not quantified here, can be expected to be higher, and in some cases very much higher than the upper bound figures below. This applies above all to the ecosystem services as part of the social costs (see Box 8 below), and to the non-use values of soil. The latter were not assessed in this study as they have rarely ever been quantified in economic terms.
- The figures reported above are ***annual costs***. In principle, they could be summed up over time using discounting procedures. As it is disputed whether discounting can be applied to soil, and at what rate, this was not done. To illustrate the effect of discounting, some calculations for the case of erosion are presented in chapter 5.5 below.

Table 29: Overview of the Total Annual Cost of Soil Degradation (in million €$_{2003}$)

		Erosion			Contamination			Salinisation	
		LB	Mean	UB	LB	Mean	UB	LB	UB
On-site costs	PC	40	588	860	96	192	289	115	278
	MC	0	222	2,272	2,187	6,658	41,234	243*	2,005*
Off-site costs	SC	680	6,676	13,139	2,283	17,126	207,615	43	43
	DC	0	2,010	2,010	482	965	1,447	-	-
Total		**720**	**9,496**	**18,281**	**5,049**	**24,941**	**250,585**	**158**	**321**

LB = lower bound, UB = upper bound (for those impacts that were quantified at all).

* The MC for salinisation are not included in the total.

Box 8: The Value of Soil Ecosystem Services

Some ecosystem services were quantified in this study, and are included in the calculations above - e.g. the role of soil for flood protection in the erosion chapter, see 4.1.2.5. However other such services, like the function of soil as a carbon pool in the global carbon cycle, could not be quantified in a consistent way for all of Europe. However, a broad indication can be derived e.g. from Hartridge and Pearce (2001). They estimate that in the UK, 7.6 million tons of carbon are released annually from cultivated soils, drained peatlands and fenlands, through peat extraction and through the transport of eroded soil to the sea (subtracting from this sequestration in forest soils, set-aside soils and undrained peatlands). Valued at £$_{1998}$ 29.80 (€$_{2003}$ 63) per ton, the total climate change impact of organic matter released from soils in the UK would amount to £$_{1998}$ 226.5 million (€$_{2003}$ 361 million) p.a. This figure is substantially higher than the £$_{1998}$ 20.5 million (€$_{2003}$ 32.7 million) figure that Hartridge and Pearce report for the annual impact of erosion (on- and off-site) in the UK. This finding also holds if a lower price for the climate change impact per ton is applied.

In a similar estimation, Pretty et al. (2000) arrive at comparable results. They value the economic impacts of soil organic matter loss in the UK at GB£ 82.3 m p.a. (€$_{2003}$ 143.3 m). The authors mainly considered the climate change impact of organic matter released from soil. In this study, the results by Pretty et al. were incorporated into the calculations for the cost of erosion (see chapter 4.1.2.5). Based on their results, about a third of the total upper-bound estimate for the social cost of erosion (SC) would be due to organic matter loss (€$_{2003}$ 6.2 billion of the total of 17.4 billion).

Along the same lines, Balmford et al. (2002) have reviewed the evidence on the economic value of different ecosystems. They provide evidence from five different ecosystems that have been converted to human use (unfortunately none of them from Europe). For all the ecosystems they consider, they find that the total benefits from conversion are actually negative. For the case of a Canadian wetland, the total economic value actually decreases by more than 40% as a consequence of conversion (from US$ 8800 to US$ 3700 / ha *y), as the loss of non-marketed services formerly provided by the wetland is not outweighed by the marginal benefits of conversion. This finding holds despite the fact that some particularly valuable ecosystem services, such as nutrient cycling and the provision of cultural values, were not considered due to a lack of data.

While these results are only indicative, they underline the importance of considering wider environmental and social benefits of soil uses, and show that a focus on the immediate soil uses (such as agriculture) can be misleading.

5.4 Interpretation of the Results

Bearing in mind the caveats explained above, some conclusions can be drawn:

- On an aggregated level, the **private, on-site costs of soil degradation** (usually suffered by land users) will not be a major cause of concern in many cases. As reported in Table 14, even the upper-bound estimate of the annual private costs of soil erosion does not exceed 0.5 % of the agricultural gross value added in the countries included in the plot data base. For the case of soil salinisation, the estimated private costs are only significant in Hungary, where the impacts could lie between 3.3 % and 6.3 % of agricultural gross value added (see also Table 27). In Spain, with estimates ranging from 0.2 % to 0.6 % of the national agricultural gross value added, the estimated private costs are manageable. However, this is also due to the angle of this study, which has focussed only on national or European averages, masking the fact that soil degradation causes considerable private on-site costs in certain regions. It should also be borne in mind that the impacts of soil degradation will often be cumulative and, in most instances, irreversible. Hence, while the costs may appear negligible on a year-to-year basis, they can become substantial when added up over a longer time.

- The **social, off-site costs of soil degradation** (covered by society) are far more substantial in most cases. For example, in the case of erosion, the cost estimates ranges from € 1.8 billion to € 14.3 billion p.a., which corresponds to 1.1 % to 8 % of agricultural gross value added for the thirteen countries covered (see Table 14). The off-site costs exceed the on-site costs by a factor of seven (for the upper bound estimate) up to a factor of seventeen (for the lower bound estimate). With regard to contamination, the situation is more complex, as off-site effects are not always present. Consequently, the extrapolation arrives at a situation where off-site-effects may be larger than or equal to on-site effects. For large contaminated sites located in populated areas, such as the French contamination case study in Volume II of this report, the off-site costs may well exceed on-site costs by a factor of five or more. The general finding that off-site costs will often surpass on-site costs holds despite the fact that off-site costs are more difficult to delineate and quantify. This difficulty applies to all types of soil degradation, and in particular to subsets like the impact on ecosystem services and on non-use values of soil.

The bulk of the costs of soil degradation will thus not be felt by the people causing it. Instead, the majority of impacts occurs off-site, affecting neighbours, downstream water users, or other ecosystems. Thus, if the focus of the analysis shifts from the individual plot or the farm level to include regional, national or even global off-site effects of soil degradation, the estimated potential impact increases rapidly. At the same time, whereas on-site effects are described fairly well in the literature, off-site effects are subject to more uncertainty. If the relevant off-site impacts, including non-use values and ecosystem services, could be quantified more comprehensively, the imbalance between on-site and off-site impacts would be even more pronounced.

On the whole, the analysis has shown that the inherent complexity of soil functions and their degradation, and the interdependencies between different soil degradation processes are difficult to grasp in an economic valuation study. To adequately account for these factors would require far more data in far greater detail than is currently available, both from the economic and from the soil scientific perspective.

5.5 Intertemporal Aspects of Valuing Soil Degradation

The cost figures presented in this report refer to the current, annual costs of soil degradation. As discussed in Box 2, the standard economic procedure used to extrapolate such impacts into the future is to discount future costs to the present, in order to obtain the net present value of the future cost of soil degradation. However, this procedure is not unproblematic:

- It has been questioned whether discounting future costs should be applied at all in the context of soil degradation, and which discount rate should be applied, given that soil is an essentially non-renewable and often irreplaceable asset (see e.g. Young (1998)).[29]

[29] It is generally difficult to reflect the aspect of irreversibility and long-term damage in an economic valuation. The standard economic approach of discounting future costs and benefits conflicts with the feeling that a unique and irreplaceable resource should be protected in its own right. In principle, such motivations be reflected in the non-use value. However, this non-use value is difficult to quantify.

- Furthermore, an extrapolation of soil degradation into the future would also need to reflect the development of main pressures on soil over time, such as land use changes, urbanisation, demographic change and climate change, in order to anticipate future trends in soil degradation. However, such information is not available at this stage.

Due to these methodological doubts, the current study did not provide an intertemporal calculation of the anticipated future costs of soil degradation. For the sake of illustration, Table 30 and Table 31 below present the Net Present Value (NPV) of the cost of erosion (intermediate estimate) for discount rates between 0% (no discounting) and 6%.

Table 30 demonstrates the effect of the discount rate on the results of the analysis if impacts are summed up over a 50-year period. At a 4% interest rate, the estimated impact of erosion over the next 50 years amounts to € 204 billion. This equals 43% of the accumulated impacts over 50 years if discounting is not applied.

Table 30: NPV$_{50}$ of the Cost of Erosion (million €$_{2003}$)

		Net present value per 50 years at interest rate of			
Cost category	**annual cost**	**0%**	**1%**	**4%**	**6%**
PC	588	29,384	23,035	12,625	9,263
MC	222	11,110	8,709	4,773	3,502
SC	6,676	333,824	261,692	143,425	105,234
DC	2,010	100,519	78,799	43,188	31,687
SUM	**9,498**	**474,837**	**372,235**	**204,011**	**149,686**

Note: Cost estimates based on the intermediate estimate for the cost of erosion

Table 31 shows that the effect of the discount rate is even more pronounced for the perpetual NPV. In this case, impacts are summed up over an indefinite period, with a cut-off point at 500 years for the 0%-interest rate, as suggested by Young (1998). In this case, the estimated net present value at a 4% discount rate sums up to € 237 billion, which is only $^{1}/_{15}$ (or 6,7%) of the figure that is obtained for a 500-year period without discounting.

Table 31: Perpetual NPV of the Cost of Erosion (million €$_{2003}$)

		Perpetual net present value at interest rate of			
Cost category	**annual cost**	**0%**	**1%**	**4%**	**6%**
PC	588	293,845	58,769	14,692	9,795
MC	222	111,097	22,219	5,555	3,703
SC	6,676	3,338,241	667,648	166,912	111,275
DC	2,010	1,005,194	201,039	50,260	33,506
SUM	**9,497**	**4,748,378**	**949,676**	**237,419**	**158,279**

Note: Cost estimates based on the intermediate estimate for the cost of erosion. For the 0% interest rate, a cut-off point of 500 years was assumed.

6 Policy Recommendations and Further Research Needs

This study has been the first in Europe to assess the economic dimension of soil degradation in a comprehensive way, across different countries and for different soil threats. The results of the study should not be seen as an exact quantification of all impacts, but rather as a way to assess the dimension of the problem of soil degradation from a different perspective. Still, many impacts need to be explored further before more definite conclusions can be drawn.

To date, it is clear that in many instances, the impacts that were not quantified in this study will exceed those that were quantified. Consequently, the upper bound figures presented here are only the upper limit for the quantifiable impacts, whereas the real impact of soil degradation will be much higher. In line with the precautionary principle, policy recommendations need to reflect not only the quantifiable impacts, but also take into account those impacts that could not be assessed in monetary terms.

6.1 Policy Recommendations

Irrespective of these caveats, this study has demonstrated that the economic impacts of current soil degradation trends in Europe are substantial, and give cause to concern. Even though many impacts cannot be quantified in monetary terms at this stage, the estimated costs presented are substantial, running into the order of several billion Euro per year. In this sense, the added value of the current study has shed some light on the magnitude of the problem, as well as the distribution between on-site and off-site costs.

- The private, on-site costs of soil degradation are suffered by soil users. For erosion or salinisation, they range between 0.5 and 2% of agricultural gross value added. While significant, these costs will generally not be a major concern in the short run. However, on the local scale, impacts will be more substantial for the affected areas. Also, impacts will be felt more strongly over time, as they are cumulative and mutually reinforcing.
- By contrast, the social, off-site costs of soil degradation are more substantial. For the different estimates presented in this study, off-site costs exceed the on-site costs by a factor of 7 to 10, despite the fact that a large part of the off-site costs could not be quantified. The off-site impacts are generally covered by society: as externalities, they are not reflected in the decision-making framework of land owner and soil users.

These discrepancies underline the economic rationale for an ambitious soil protection policy. In the short term, the private, on-site costs are mostly moderate. Even where they are significant, the fact that the soil user is often not the same as the soil owner means that the soil user has no incentive to protect the soil beyond the rental term, leading to unsustainable soil use. The off-site, social costs are substantial, but are covered neither by the polluters nor by insurers, so that there are few incentives for changed behaviour. In line with the polluter-pays-principle, **policy solutions** are therefore necessary to **change these incentives**. By internalising the external costs of soil degradation, off-site impacts can be better integrated into the decision-making and the behaviour of soil users. In principle, this can be done through taxation, through behavioural codes, or through conditionality for subsidy payments.

In practice, however, it may be problematic to relate a specific, localised off-site impact to an individual soil use. For soil contamination, significant time lags may exist between the contamination itself and the detection of off-site impacts. For salinisation or erosion, the relative contribution of individual soil uses to the occurrence of off-site impacts is often difficult to establish. To address this, **more effective and unified soil monitoring** is required. Soil monitoring systems need to be designed in such a way that the link to the assessment of socio-economic impacts is easily made. In particular, soil monitoring can be used to support the use of political instruments aimed at internalising external costs.

To some degree, the internalisation of off-site effects of soil degradation can be achieved through better **integration of soil protection requirements** into other policy areas.

- For soil degradation caused by agricultural soil uses, the most suitable instrument to address off-site effects is through the use of the **cross-compliance mechanism** established under the Common Agricultural Policy. Here, it is necessary to better integrate off-site effects into the definition, guidance and the practical implementation of “good agricultural and environmental conditions” and “good agricultural practice”. Next to the cross-compliance mechanism, voluntary approaches such as **cooperative agreements** could also be effective, as demonstrated by experiences in the area of diffuse water pollution (see e.g. Heinz et al. 2002).
- Soil protection requirements should also be better integrated into the **implementation of the Water Framework Directive**. By 2009, the WFD mandates the establishment of programmes of measures, which should achieve the good ecological status for water bodies in the most cost-effective way. Currently, off-site effects of soil degradation are among the main pressures that prevent water bodies from reaching good ecological status, e.g. in the case of erosion-induced water pollution or groundwater contamination from contaminated land. Where impacts on a water body can be related to soil degradation, the most cost-effective way of addressing them could include soil stabilisation, decontamination and improved agricultural practices.
- Furthermore, there is a clear link between soil protection and **flood risk management**. In the developing European approach to flood risk management, flood prevention measures are becoming increasingly relevant to support and complement structural / technical flood protection measures (see e.g. Dworak and Hansen 2003). In order to prevent or limit floods and to reduce flow velocity, the capacity of soils to absorb and retain rainwater in upstream areas needs to be enhanced. This can be achieved e.g. through measures that reverse or limit soil compaction and soil sealing. Soil protection and land use policies can thus make a significant contribution to flood risk management. In view of the substantial economic damage caused by flooding events, such measures offer themselves as a relatively inexpensive contribution to flood prevention.
- In the area of **climate change,** soil protection needs to play a double role: first, maintaining healthy soils and the build-up of organic matter can enhance the role of soil as a sink for atmospheric CO_2. By contrast, soil degradation will lead to the release of carbon from soils. Furthermore, soil protection will also be key to adaptation strategies, as the resilience of ecosystems to adapt to the changing climate depends not least on vital and multifunctional soils. The policy objective must therefore be to stabilise and, where possible, increase the level of soil organic matter.
- For the area of **land use and spatial planning**, the planning of industrial, residential and commercial development needs to take more account of soil properties. In order to minimise the cost of soil degradation, it is not only necessary to protect the most vulnerable soils, but also to identify soils that are more suitable for polluting or degrading activities, and to concentrate such activities on such soils. For the remediation of contaminated land, the objective has to be to minimise new contamination and prevent

accidental pollution, to decontaminated existing contaminated sites as far as possible, and to limit the affected area by preventing the spread of mobile pollutants.

Other policy areas where soil protection requirements need to be better integrated are internal market policies, chemicals policies and transport. The issue of demolition waste and construction material is a particular example of this, as it lies at the interface between internal market policies, waste policies and soil protection.

The enhanced integration of soil protection requirements should be supported through the nascent European Thematic Strategy on Soil Protection. For any of the policy areas mentioned above, the integration of soil policy requirements will depend on clear definitions and indicators for soil quality, as well as specified objectives for soil protection. Delivering such definitions, indicators and objectives should therefore be one main focus of the Thematic Strategy.

6.2 Research Needs

The current study should thus be regarded as a first step, which needs to be developed and refined further. In addition to the need for a coherent soil monitoring system identified above, socio-economic **research needs** concern four issues in particular.

- The concept of **ecosystem services** that captures the interactions between soils and other parts of the ecosphere. Since soil is closely related to the natural processes taking place in the hydrosphere, the atmosphere, the lithosphere and the biosphere. Therefore the degradation of soils will have a direct impact on the functioning of these other compartments. These interactions are of particular relevance in the context of climate change. Global warming will not only affect soil ecosystems through increased temperatures and higher levels of precipitation; at the same time, soil as the third largest global carbon pool also plays a key role for the stability of the global climate system itself. Through such interactions, soil provides plenty of different ecosystem services, not all of which are sufficiently understood. While many ecosystem services could not be assessed economically in the course of this study, there is some evidence that adding ecosystem services into the equation can affect the judgement on the economic viability of different land uses. In particular, the value of lost ecosystem services may far outweigh the short-term benefits of intensive land use, whereas sustainable soil management practices can enhance the ecosystem services provided by soils (see e.g. Balmford et al., 2002).
- A second category that merits closer inspection are the **non-use values** of soil. Soil as a non-renewable and non-replicable resource has been the fundament of human development since the very beginnings of civilisation, and bears manifold cultural and spiritual connotations. Soil therefore needs to be protected both in its own right, and as an asset for future generations. From an economic perspective, such considerations would form part of the non-use value of soil. However, this non-use value has barely been researched at all, safe for a few Australian estimates.
- In terms of **different soil threats**, several types of soil degradation could not be assessed comprehensively. This was either due to a lack of economic data, or due to the absence of comprehensive soil data on the European level, or both. For threats such as the loss of soil organic matter, the loss of soil biodiversity, soil sealing and soil compaction, more **primary studies** are needed in order to assess the economic impacts caused by these threats.

- A fourth research challenge concerns the **intertemporal valuation of soil degradation**. This concerns not only the choice of the appropriate discount rate, but more importantly the questions of how to deal with irreversibility, and how to predict and incorporate the resilience of soils to increasing pressures. To move ahead in this regard, a "baseline scenario" for soil degradation would be necessary in order to assess how pressures on soil are likely to develop over time, how this will affect soil quality and resilience, and what impact this will have on soil users.

To address these questions, research projects and networks would need to be established under future calls of the 6^{th} and in the 7^{th} Framework Programme on Research and Development. These should include both basic research and policy oriented research, with the aim of building up and extending the European knowledge and data base.

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Annex 1: Assessment of Other Soil Threats

A-1.1 Decline in Organic Matter

A-1.1.1 Situation

At the European scale, three different types of organic matter loss can be distinguished:

(i) intensive agriculture and progressive depletion of organic matter content in middle-latitude regions (e.g., France, the Netherlands, Germany),

(ii) historic and extensive organic matter loss due to climate change and desertification in Southern Europe,

(iii) peat extraction in Northern Europe (Scandinavia, Ireland).

Organic matter decline in the two last forms mainly occurs in agricultural areas, as the agricultural activities greatly influence the stock of organic matter in soil (entry and release). The WG on Organic Matter and Biodiversity, TG 7 argues that ploughing of agricultural soils is a major contributor to the decrease of soil organic matter. Also, tillage may reduce organic matter concentrations and increase organic matter turnover rates.

The WG on Organic Matter and Biodiversity, TG 7 also notes that it is difficult to estimate the impacts of agricultural land mismanagement on European agricultural production. Nevertheless, it is evident that the overall fertility of soil largely depends on organic matter content. If the soil organic matter content falls below a certain threshold (between 2 and 3.4%, depending on the soil type), the levels of production are bound to fall. At the same time, the estimation of costs is hindered because the impacts of decline in organic matter are analogous to the impacts of other soil degradation types, such as soil erosion and loss of soil biodiversity, and often occur in conjunction with these.

The WG on Organic Matter, TG 2 (2004, p. 3) asserts a “serious lack of georeferenced, measured, harmonised data on soil organic carbon available at the European level, from systematic sampling programmes”. The data that exists has mainly been gathered on the national scale, thus not comprehensive geographically. Decline in organic matter is of particular concern in southern Europe, including Greece, Italy and France, but also in England and Wales (see also Jones et al., 2004 in Volume II of this report).

Peat extraction is a special form of decline in soil organic matter. Data on the extent of peat extraction is scarce at the European level (Joosten and Clarke, 2002). The data are mainly available on industrial peat extraction for horticulture and energy production. However, little is known about the amounts of non-industrial peat extraction that remains an important local energy source. Peat is an important local or regional energy source in Finland, Ireland, Sweden and the Baltic States. Finland, Germany, Ireland, Sweden and the UK provide the domestic horticultural enterprises with peat resources.

A-1.1.2 Method

Decline in soil organic matter and especially peat extraction have specific impacts on soil and other environmental media that could be grouped under the different cost categories (as established by Volume I of this report, chapter 4.6) (see Table 32).

Table 32: Impacts of decline in soil organic matter sorted into cost categories

On-site costs	PC	▪ Soil organic matter plays a central role in maintaining key soil functions, such as keeping soil structure, retaining water and as a nutrient reserve, and is thus an essential determinant of soil fertility. As a result, the decline in organic matter impacts directly the fertility of soil and, hence, the agricultural productivity. Costs are associated with losses of yield.
	MC	▪ Measures to restore soil organic matter (were possible), costs of additional fertiliser application.
Off-site costs	SC	▪ Soil organic matter content determines its capacity to absorb pollutants. When biological activity of soil is reduced, it is more prone to leaching, affecting ground and surface water quality. The decline in organic matter impacts directly ground and surface water pollution and indirectly the soil contamination, causing additional costs.
		▪ As C is a major component of soil organic matter, soil is one of the biggest pools of C, which in turn plays a major role in the global C cycle.[30] The decline in organic matter impacts indirectly the function of the soil as storage (sequestration) of organic C and the mitigation of atmospheric increase in CO_2. Effects of climate change arise through the reduced capacity of degraded soil to store C. Cost are associated with the suffered climate change impacts.
	DC	▪ Costs for defensive measures against climate change impacts.
	NC	▪ A decline in organic matter leads directly to a loss of biological activity and biological diversity of soil. This affects soil fertility (PC), but also the genetic resources present in soil (SC / NC)
		▪ Damage to landscape features in extreme case of organic matter loss.

[30] Research indicates that approximately 2 Gt of C are captured (sequestered) in soil organic matter annually (Lal, 2000; European Commission, 2002).

Box 9: Peat Extraction as a Special Case of Organic Matter Loss

The extraction of peat from mires and peatlands is a special form of organic matter loss. Resource extraction is a direct impact of peat mining **(PC)**. While peat mining leads to the loss of organic matter in one place, the extracted peat is used elsewhere, for example in horticulture and agriculture, but also for remediation of degraded soils and as a topsoil replacement for the regeneration of former open-cast mining sites. Moreover, peatlands perform a number of functions in situ. If peat is extracted, the ecological functions of mires are disturbed or destroyed. These include the role of peat for climate local regulation, with peats as one of the major carbon sinks in Europe,[31] as well as biodiversity maintenance and the storage and purification of water **(SC)**. Peat extraction, in addition, causes indirect impacts such as loss of cultural and heritage values of peatlands **(NC)**. Peatlands and mires have also important social functions, for instance they are popular for recreational activities like hiking, hillwalking and birdwatching **(SC / NC)**.

While the economic impacts of organic matter loss through peat mining can be described, their quantification is obstructed by the lack of data. However, one aspect that has been researched in the past is the role of peat in the carbon cycle. If peat is extracted and used as a fuel, not only does it lose its function as a carbon sink, its burning also leads to CO_2 emissions that add to global climate change and thereby cause additional costs. The amount of carbon bound in peatlands can be substantial: Pretty et al. (2000) estimate that of the 21.78 billion tonnes of carbon stored in soils in the UK, 16.4 billion tonnes (some 75%) are contained in Scottish peats alone. Hartridge and Pearce (2001) provide some evidence for the UK on how much carbon is released annually through peat mining. They argue that, out of total annual carbon losses from soils of 9.7 million tons, only 200,000 tons are due to peat mining, representing only 2% of the total. Another 800,000 tons of carbon are released through the draining of peatlands and fenlands. However, the vast majority of carbon losses from soils are due to agricultural practices.

In terms of economic analysis, peat extraction for commercial purposes is a particular case, as the extraction of organic matter is done intentionally – as opposed to other cases of organic matter loss, where it occurs as an unwanted side-effect. Therefore, peat extraction is not a typical form of soil degradation. This has some implications for the economic assessment of peat extraction. On the one hand, peat extraction is a type of organic matter loss in soil. On the other hand, peat is also a resource, and hence has a commercial value e.g. as a fuel for heating and energy generation. However, as peat is essentially non-renewable in the timeframe of an economic analysis, it is doubtful whether the market price of peat indeed reflects adequately the value of ecological and other functions that are lost through peat extraction.

Therefore, in order to reflect the economic impacts of organic matter loss through peat extraction, the ecological functions and values of peatlands (in-situ value) have to be considered, as well as the commercial value of peat as a resource (extractive value). However, this point has not been addressed coherently in the literature. Some insights on this issue can be found, however, in the case study on organic matter loss that was prepared as part of this project (see Volume II of this report).

[31] According to the WG on Organic Matter, TG 2 (figure 2.22), 49% of the organic carbon stock in topsoils in the EU-25 is located in three countries: Sweden (20.2%), Finland (18.3%) and the UK (10.4%). The vast majority of carbon storage in these countries occurs in peat soils – in Finland, 80% of the total carbon storage occurs in peatlands, against only 5% in agricultural soils. In the UK, this share is estimated at 75%.

A-1.1.3 Results

The economic effects of decline in organic matter are covered only in occasional studies, or are not quantified at all. The lack of studies quantifying the economic impact of organic matter loss was also recognised by the WG on Organic Matter and Biodiversity, TG 7 (2004). The WG points out that several regions in Southern Europe have reached a critical level of organic matter, below which agricultural production might decrease sharply.

In one of the rare quantitative studies, Pretty et al. (2000) estimated the economic impacts of organic matter loss in the UK at GB£ 82.3 m p.a. (€$_{2003}$ 143.3 m p.a.). To do this, the authors mainly considered the climate change impact of organic matter that is released from the soil. Within this study, the results by Pretty et al. were incorporated into the calculations for the cost of erosion. Based on their results, about a third of the total upper-bound estimate for the social cost of erosion (SC) would be due to organic matter loss (€$_{2003}$ 6.2 billion of the total of 17.4 billion). As discussed in footnote 10 above, Pretty et al. argue that part of the organic matter loss may be caused or enhanced by erosion; however this link is not quantified.

Furthermore, Hartridge and Pearce (2001) estimate that in the UK, 7.6 million tons of carbon are released annually from cultivated soils, drained peatlands and fenlands, through peat extraction and through the transport of eroded soil to the sea (subtracting from this sequestration in forest soils, set-aside soils and undrained peatlands). Valued at £$_{1998}$ 29.80 (€$_{2003}$ 63) per ton, the total climate change impact of organic matter released from soils in the UK would amount to £$_{1998}$ 226.5 million (€$_{2003}$ 361 million) p.a.

Hogg et al. (2002) describe some measures to promote accumulation of organic matter in the soil that have taken place in Italy, under the scope of Rural Development Plans (2000-2006). Subsidies are given to farmers who use organic fertilisers such as compost in order to increase levels of organic matter. The measures, at the same time, are deemed to combat climate change and desertification as well as enhance the overall productivity of farmlands. For example, for the last couple of years, the region Emilia Romagna has been paying some 130 €/ha to use compost and promote a build-up of soil organic carbon in depleted soils.

The decline in soil organic matter induces other soil degradation types, e.g. soil erosion, and vice versa. The estimation of costs is therefore difficult. Moreover, the impacts of floods and landslides, i.e. the loss of soil “buffer” functions, that are strictly linked to organic matter, should be taken into account.

A-1.1.4 Interpretation

The literature review, the case studies and the database research indicate that a Europe-wide assessment of the economic impacts of organic matter losses is currently not possible.

- The assessment of soil organic matter loss on a European level is severely limited by the lack of data on the extent and the spatial distribution of organic matter loss in Europe. While some data exists on the organic matter *content* of soils, there no consistent Europe-wide data on the organic matter *losses*. The availability of data is also limited by the fact that there is no clear categorisation for different types of organic matter, or for the impacts of soil organic matter loss.
- Furthermore, the impact of organic matter loss on the productivity of soils is much less researched than e.g. in the case of erosion, rendering an economic assessment more difficult. Since organic matter loss and soil erosion will often occur in conjunction, it can be assumed that part of the on-site economic damage ascribed to erosion (see chapter 4.1) is in fact related to the loss of soil organic matter.

- For the off-site effects of soil organic matter loss, there is some evidence that the climate change impact of carbon released from soils is substantial. For the UK, Pretty et al. (2000) and by Hartridge and Pearce (2001) estimate the cost of these impacts to lie between €$_{2003}$ 143 million and €$_{2003}$ 361 million. This is considerably higher than the total off-site cost of erosion that Hartridge and Pearce estimate for the UK (see also Box 8). However, at this stage, the lack of Europe-wide data on the extent of organic matter loss prevents us from carrying out such calculations on the European level. A very crude estimation, based on the total agricultural area in the UK and the EU-25, would imply that the total cost in Europe could be about 10 times at high (€ 1.4 billion to € 3.6 billion).[32]
- Finally, the economic assessment of organic matter loss is complicated by the fact that different types of organic matter loss occur in Europe (i.e. peat mining in the North of Europe, agricultural organic matter loss in the middle latitudes, and desertification in Southern Europe). Each of these has its own characteristics; consequently, a different economic approach is called for in the different cases. In the case of peat mining, the analysis has to recognise the peculiarity that the loss of organic matter is inextricably linked to an economic activity – in contrast to other cases, where degradation occurs as an unintended by-product.

[32] The total agricultural area of the UK, 15.7 m ha, represents 9.3% of the agricultural area of the EU-25. The total carbon stock of the UK, 7.1 Gt, equals 10.4% of the total carbon stock for the EU-25. As comprehensive data on organic matter losses in the EU-25 is not available, the approximation based on the agricultural area and the total carbon stock represents a best guess.

A-1.2 Floods and landslides

A-1.2.1 Situation

Floods and landslides are unique in that they are both a cause and a consequence of soil degradation. The Commission Communication "Towards a Thematic Strategy for Soil Protection" notes that "Floods and landslides are not a threat to soils in the same manner as the threats already listed. However, floods can, in some cases, result in part from soil not performing its role of controlling the water cycle due to compaction or sealing. They may also be favoured by erosion often caused by deforestation or by abandonment of land.“ Along these lines, it could be argued that floods form part of the off-site effects of soil erosion, soil sealing or soil compaction.

At the same time, floods and landslides can also be a cause of soil degradation. Floods and landslides directly impact soil by washing out the fertile topsoil, leading to a loss of productive soil and hence a decrease in crop yield as well as loss of soil resources. Floods can be a cause of diffuse contamination if contaminated sediment is washed out and deposited on floodplains.

However, the empirical assessment of these interactions has not been researched extensively. While it is possible to assess the aggregated cost of flooding events in Europe, it is highly difficult to establish which part of the damage can be traced back to soil degradation in upstream areas. Likewise, it is difficult to establish which part of the total cost takes the form of environmental impacts caused by the deposition of contaminated sediment.

Floods and landslides are characteristic to river basins and coastal areas. In addition, landslides mainly occur in mountainous and hilly regions. EEA data (2000c) indicates that the Alpine and Mediterranean regions are the regions most affected by floods and landslides, although data is only available for Italy. At the same time, the WG on Erosion (2004) claims that flooding is a problem for large parts of Europe, in particular Southern and Central Europe. Central Europe is indicated as having a unique situation because regions that are affected by extreme flood events suffer from severe drought later in the same year. In addition, as regards the situation on landslides, WG on Erosion (2004) claims that this phenomenon is being increasingly recognised as a primary hazard, and is analysed in several European countries.

A-1.2.2 Method

Floods and landslides are not typical soil degradation forms. Both phenomena occur accidentally and have a specific pattern of impacts depending on a scale, place and reason. Also, as noted above, they can be both a cause and a consequence of soil degradation. Nevertheless, characteristic impacts can also be defined and attached to specific cost categories (see Table 33).

If floods and landslides are considered as a consequence of soil degradation (through changed runoff dynamics and reduced water retention capacity), the associated impacts should be considered as off-site impacts of soil degradation in the upstream areas. Consequently, these impacts were all sorted into the SC category in the table below.

Table 33: Impacts of floods and landslides sorted into cost categories

		Impact of floods on soil	Impact of soil degradation on floods
On-site costs	PC	▪ Floods and landslides impact soil by washing out fertile topsoil, leading to a loss of productive soil and hence a decrease in crop yield.	
	MC	▪ Cost of replacing topsoil or stabilising soil, where applicable.	
Off-site costs	SC	▪ Contamination of land through rupture of underground pipelines, dislocation of storage tanks, overflow of toxic waste sites or the release of chemicals stored at ground level. ▪ Pollution distribution with sediments and consequent water and soil contamination, associated cost of remediation and clean-up.	▪ Impacts on human lives and wellbeing, ▪ Damage or loss of property (buildings, goods and infrastructure). ▪ Indirect effects, through the interruption of transport routes and production losses ▪ Ecological damage from floods ▪ Anxiety and uncertainty of people exposed to floods and landslides.
	DC	▪ Defensive expenditure to prevent soil contamination, and to maintain soil in place (flood-proof storage tanks, soil stabilisation measures).	
	NC	▪ Deposition of contaminated sediment can be harmful to wetland ecosystems in floodplains, which may have a high non-use value due to their role in sustaining biodiversity.	

A-1.2.3 Results

While a number of studies have calculated the economic effects of floods, the data situation is somewhat poorer for landslides. Generally, the focus of such studies tends to be on the direct damage caused by floods and landslides, e.g. through the destruction of roads, bridges, houses or other private property. The indirect economic effects, such as the economic damages induced by disrupted transport routes, are less easily estimated. In the following, some results will be presented for the cost of floods and for the cost of landslides.

Estimation of the Cost of Floods

The following examples of the cost of floods were already discussed in the literature review:

- Munich Re (2002) present costs figures for Rhone River in France and Danube & Elbe in Austria, the Czech Republic and Germany. The cost of damages of September 2002 flood in Rhone River is assessed to be equal to € 1.2 bn **(SC)**; and the December 2003 flood was estimated at € 1.5 bn **(SC)**. The damage cost of the August 2002 floods in Danube & Elbe rivers was assessed at € 18.5 bn for Austria, the Czech Republic and Germany **(SC)**. In all three cases, the share of the damage that can be related to soil degradation was not quantified (Volume I of this report, Table 13).
- Clark, Haverkamp & Chapman (1985) present the cost of flooding as part of the off-site damages of erosion in USA at US$$_{1980}$ 490 m (€$_{1999}$ 1005 m) **(SC)**. Eastwood, Krausse, and Alexander (2000) presents cost of increased flood severity as the off-site costs of erosion in New Zealand at NZ$$_{1998}$ 14. 0 m p. a. (€$_{1999}$ 9.6 m p. a.) **(SC)** (Volume I of this report, Table 12).

In addition, Table 34 and Table 35 below present some evidence of the total and annual cost of flooding events in the EU. On a country-by-country basis, the tables present casualties inflicted by flooding events, the number of people affected (i.e. people requiring immediate assistance and basic survival needs such as food, water, shelter, sanitation and immediate medical assistance), and estimates of the costs of the flooding events. The reported economic impacts depend on the methodology applied, but would normally include direct impacts (PC category – such as damage to infrastructure, crops and housing) as well as indirect impacts (SC category – e.g. loss of revenues, unemployment, market destabilisation). This information is presented as a total (for all flooding events covered in the national statistics), and as a lower-bound and upper-bound average. Since the database only presents the sum of direct and indirect impacts, the table below presents PC and SC as a sum.

The information in the table has been distilled from the OFDA / CRED International Disaster Database of the Université Catholique de Louvain, Belgium, which lists a total of 196 flooding events from European Countries. The average values reported in the OFDA/CRED database are likely to be underestimates, as information on the economic impact is not reported for all events: Of the 196 flooding events covered in the database, only 87 contain economic information. However, the average figures in the OFDA/CRED database are calculated for all events, and not only for those that report economic information. Therefore, the average values reported in the database were used as lower-bound (LB) estimates for the average cost of flooding events. In addition, an upper-bound (UB) estimate was calculated as the average of only those cases that included information on the economic impacts.

Table 34: Incidence and Costs of Floods in Europe

	No. of events	Casualties (total)	Casualties (av. / event)	Affected total)	Affected (av. / event)	Cost (total, million €) PC + SC	Cost per event (LB av., million €)	Cost per event (UB av., million €)
Belgium	19	30	2	11,025	580	29.00	1.53	29.00
Germany	14	57	4	579,500	41,393	12,525.73	894.69	1,138.70
Greece	15	78	5	10,990	733	719.52	47.97	239.84
Spain	19	1,270	67	734,250	38,645	7,027.29	369.86	702.73
France	36	233	6	88,651	2,463	4,109.84	114.16	513.73

	No. of events	Casualties (total)	Casualties (av. / event)	Affected total)	Affected (av. / event)	Cost (total, million €) PC + SC	Cost per event (LB av., million €)	Cost per event (UB av., million €)
Ireland	3	3	1	3,800	1,267	38.00	12.67	38.00
Italy	30	1,189	40	1,465,650	48,855	15,374.11	512.47	854.12
Netherlands	4	2,000	500	564,000	141,000	2,133.00	533.25	533.25
Austria	7	26	4	60,000	8,571	2,128.91	304.13	709.64
Portugal	10	562	56	39,986	3,999	143.10	14.31	28.62
Sweden	2	11	6	n.a.	n.a.	175.80	87.90	87.90
UK	16	51	3	3,570	223	6,569.59	410.60	729.95
Sum EU-15	**175**	**5,510**	**31**	**3,561,422**	**20,351**	**50,973.88**	**291.28**	**679.65**
Czech Rep.	2	47	24	287,725	143,863	2,150.30	1,075.15	1,075.15
Hungary	8	309	39	144,871	18,109	414.44	51.80	103.61
Poland	7	89	13	199,000	28,429	5,103.00	729.00	1,701.00
Slovakia	4	56	14	46,057	11,514	141.35	35.34	47.12
Sum EU-25	**196**	**6,011**	**31**	**4,239,075**	**21,628**	**58,782.97**	**299.91**	**675.67**

Source: EM-DAT: The OFDA/CRED International Disaster Database www.em-dat.net – Université Catholique de Louvain, Brussels, Belgium. No data was reported for Denmark, Luxembourg, Finland, Cyprus, Malta, Estonia, Latvia, Lithuania and Slovenia. The temporal range of the data depends on the national statistical coverage and differs between 97 years (Belgium) and six years (Czech Republic).

Several points should be noted:

- Singular events can have considerable influence. For example in the case of Germany or Austria, where 70% (Germany) to 96% (Austria) of the total damage costs are due to the August 2002 Elbe flood alone. The same applies to Poland (84% due to the June 1997 floods) and the Czech Republic (93% of damage caused by the August 2002 flood). This also partly explains the high average costs reported for these countries. In Belgium, cost was reported for only one of 30 flooding events (January 1995 for Dinant and Liege).
- In most countries, the most severe impacts of flood damage have occurred in recent years. Of the total estimated damage of € 58.8 bn for the EU-25, some 45 per cent (or € 26.7 bn) have been reported since 2000 alone. Of this, € 13 bn were caused by the August 2002 floods of the Elbe and Danube rivers alone. The only exception to this are Spain and Portugal, where the reported damage has mainly occurred in the 1970s and 1980s. However, this tendency is not only due to a higher incidence of floods, but is also caused by an inherent bias in the data. Only five of the EU-25 countries report economic data on flood damage prior to 1970, and only one prior to 1950. France and Belgium have flood statistics (on occurrence, casualties and affected people) covering almost a century, but economic impacts have only been included in the last ten to twenty years.

Table 35 below presents some calculations relating the data reported in Table 34 to the time covered in the national statistics. In doing this, the reported flooding events and the associated casualties and costs have been applied uniformly across the time period covered in the national statistics, in order to give a long-term and lower-bound average. In a second adjustment step, these averages have been calculated not for all years covered in the national statistics, but only for the years in which economic information had actually been reported (beginning with the first occurrence). Compared to the unadjusted lower-bound average, these adjusted figures were then taken as upper-bound long-term average.

Table 35: Average Cost of Flooding Events

	Start year (national stats)	Earliest year w. economic info	Years covered (national stats)	No. of events	No. of events w. economic info	Events per year (av.)	Casualties / year (av.)	Cost per year (LB av., million €)	Cost per year (UB av., million €)
Belgium	1906	1995	97	19	1	0.20	0.31	0.30	3.63
Germany	1920	1920	83	14	11	0.17	0.69	150.91	150.91
Greece	1977	1977	23	15	3	0.65	3.39	31.28	27.67
Spain	1953	1962	50	19	10	0.38	25.40	140.55	171.40
France	1909	1983	94	36	8	0.38	2.48	43.72	205.49
Ireland	1983	1993	20	3	1	0.15	0.15	1.90	3.80
Italy	1905	1951	98	30	18	0.31	12.13	156.88	295.66
Netherlands	1953	1953	50	4	4	0.08	40.00	42.66	42.66
Austria	1954	1991	49	7	3	0.14	0.53	43.45	177.41
Portugal	1967	1967	36	10	5	0.28	15.61	3.98	3.98
Sweden	1976	1976	27	2	2	0.07	0.41	6.51	6.51
UK	1952	1977	51	16	9	0.31	1.00	128.82	252.68
EU-15			**678**	**175**	**75**	**3.12**	**102.10**	**719.67**	**1,341.79**
Czech Rep.	1997	1997	6	2	2	0.33	7.83	358.38	358.38
Hungary	1970	1970	33	8	4	0.24	9.36	12.56	12.56
Poland	1928	1982	75	7	3	0.09	1.19	68.04	243.00
Slovakia	1974	1998	29	4	3	0.14	1.93	4.87	28.27
EU-25			**821**	**196**	**87**	**3.93**	**122.41**	**1,194.81**	**1,984.00**

Source: EM-DAT: The OFDA/CRED International Disaster Database www.em-dat.net - Université Catholique de Louvain, Brussels, Belgium; own calculations. No data was reported for Denmark, Luxembourg, Finland, Cyprus, Malta, Estonia, Latvia, Lithuania and Slowenia.

From the analysis, the following points should be noted:

- Summing up the number of flooding events for the EU may lead to double counting if a flood affected several countries. E.g., in the case of the August 2002 floods on the Elbe and Danube rivers, the flood would appear as a separate incident in the national statistics of Austria, Germany and the Czech Republic. However, this only affects the statistics on the number of flooding events per year, but not the data on casualties or costs, as these are reported nationally.
- The incidence of flooding is distributed unevenly between different countries. Over the period covered in the respective national statistics, Greece experienced a flood every 1.5 years on average, Spain and France every 2.6 years, and roughly every three years in Italy and the Czech Republic. By contrast, Poland, Sweden and the Netherlands have experienced floods less than once in ten years. For the EU-25 countries not included above, no evidence of flooding events was reported. Added up for the EU-25, this equals 3.93 flooding events per year on average.
- The average annual cost is highest in the Czech Republic, with € 358 m p.a.; this is owed both to the high singular cost of the August 2002 floods, and to the fact that the database only reports Czech data for the last six years. The average annual costs exceed € 120 m for Germany, Italy, Spain and the UK.
- For the EU-25, the average annual costs amount to € 1.19 billion for the lower-bound estimate, and € 1.98 billion per annum for the upper-bound estimate. This figure may be somewhat skewed, as the exceptionally high figure from the Czech Republic contributes between 18 and 30% of the total to the upper-bound and the lower-bound estimate.
- However, it has to be noted that these figures are long-term average figures that cover the damage caused by floods over the last ten to fifty years (depending on the data situation in the different Member States). It is very likely that these damage figures will be higher in the future, mainly because of the impacts of a changing climate on the water cycle (see e.g. EEA 2004).
- Also, even the corrected average figures for may still be an underestimation of the actual damage: first, because the reported cost data may not be fully comprehensive for all types of damages, and secondly, because cost data has not been reported for all events.

It should be underlined that the figures above only report the total costs of flooding, but do not identify the contribution of different causative factors. Thus, while these data represent the total cost of flooding events, it has not been possible to estimate:

- How much of the total cost of flooding is due to soil degradation (erosion, compaction, sealing) in upstream areas;
- What part of the total damage takes the form of flood-induced contamination, e.g. through deposition of contaminated sediment.

Concerning the first aspect, i.e. soil degradation as a cause of erosion, there is widespread agreement among soil scientists that soil degradation (esp. erosion, sealing and compaction) will change runoff dynamics in the catchment area of a river, and may thereby increase the likelihood and the severity of floods and landslides. Unfortunately, the linkage between soil degradation processes in upstream areas and the incidence of flooding downstream is still poorly understood, therefore this connection can hardly be qualified.[33]

As a rough estimation, the Saxonian State Ministry for the Environment and Agriculture assumes that up to 10% of flood damages at the Elbe flood in 2002 was due to soil sealing and soil compaction in upstream areas (Martin Socher, personal communication, 2004). However, this number cannot simply be transferred to other cases, as the linkage between soil degradation and floods is influenced by several factors, including the frequency, duration and severity of the weather events triggering the floods, or the local topography of the flooded area. Especially in cases of extreme rainfall leading to flash floods, it is possible that the influence of soil degradation is limited: rainfall of 100 litres / m^2 or more exceeds the retention capacity of almost any soil, be it a sealed parking lot or a forest.

If, despite these reservations and limitations, the figure of 10% is assumed to be indicative of the situation in other catchments and for other flooding events, this would imply that the annual soil-related cost of flooding events could have reached up to € 200 million on average in the past (with € 120 million as a more conservative estimate). It should be noted that these figures are backward-looking, based on a long-term average of flood damage. Since there are some indications that flood damage is likely to increase in the future (e.g. EEA 2004), the long-term average data may not be suitable for an extrapolation into the future.

A different approach is to consider the area affected by erosion, and to deduce therefrom the cost of floods as an off-site effect. The calculations presented in chapter 4.1.2.5 include, inter alia, information on the erosion-induced damage through floods (see also Table 7). Based on studies by Eastwood et al. (2000) and Clark et al. (1985), the annual average cost of erosion-induced flooding was estimated at € 7.94 per ha. For the 13 countries included in the BRGM database, this amounts to a total cost of € 617 million per annum. These costs are included in the best-guess estimate of € 8.88 billion and the upper-bound estimate of € 16.29 billion in Table 13). Ideally, such an estimation would also be necessary for soil compaction and soil sealing. However, it appears that the impact of compaction and sealing on the damaged caused by flooding events has not been quantified yet.

Estimation of the Cost of Landslides

In comparison to the data coverage on floods, the data situation for landslides is much less satisfactory. The WG on Erosion presents two examples of the cost of landslides:

- In Sweden, due to erosion formed by rock falls, landslides and gullies, the damage and rebuilding costs for the Swedish society are approximately € 10 million per year. The costs cover the expenses for measures to repair and rebuild buildings, infrastructure and other constructions (WG on Erosion, 2004) **(PC / SC)**.
- In Poland, after heavy rainstorms and flood in summer of 1997, numerous landslides occurred in the Polish Carpathians. Serious damages to buildings and communication infrastructure have been continuously reported since. The rough evaluation of costs is € 10 million (WG on Erosion, 2004) **(PC / SC)**.

[33] The lack of reliable quantified data in this field was confirmed inter alia by Reinhard Schmidtke of the Bavarian Water Management Agency (Bayerisches Landesamt für Wasserwirtschaft), who is one of the leading experts in the field of flood damage assessment and valuation.

OFDA/CRED International Disaster Database of the Université Catholique de Louvain contains only twelve cases of landslides for the EU-25 countries, two thirds of which are from Italy. Of the twelve events covered in the database, quantified economic information is provided only for three cases (two Italian and one Swedish case). The cases from Austria and the UK only provide information on the casualties, but not on the economic damage suffered.

In addition, none of the landslides reported in the database are more recent than 1987, which makes it hard to assess on this basis whether landslides should still be seen as a major threat, and whether their incidence has increased in tendency.

Table 36 below presents the quantified evidence on the incidence and costs of landslides that could be inferred from the OFDA/CRED database. It should be noted that of the 12 events of flooding documented in the database for European Countries, only three contain quantified evidence of the economic damage. If such data were available for other landslides as well, the total cost would be substantially higher.

Table 36: Incidences and Costs of Landslides in Europe in €$_{2003}$

	No. of events	casualties (total)	casualties (av.)	affected (total)	affected (av)	cost (total, €)	cost (av., €)
Austria	2	43	22	-	-	-	-
Italy	8	1,387	173	10,100	1,263	1,733,000,000	866,500,000*
Sweden	1	13	13	50	50	20,000,000	20,000,000
UK	1	140	140	-	-	-	-
Sum	**12**	**1,583**	**132**	**10,150**	**846**	**1.753.000,000**	**584,333,333****

Source: EM-DAT: The OFDA/CRED International Disaster Database www.em-dat.net - Université Catholique de Louvain, Brussels, Belgium.

* average based on two out of eight cases, for which there is quantified economic data.

** the average figure (average per event) is based on the three cases of landslides where quantitative data on economic impacts was available (Valtelina / Italy, Juli 1987, €$_{2003}$ 669 m damage; Ancona / Italy, December 1982, €$_{2003}$ 1064 m damage; Gothenburg / Sweden, December 1977, €$_{2003}$ 20 m damage).

A-1.2.4 Interpretation

In contrast to other threats to soil, floods and landslides are natural hazards, meaning they are singular and often catastrophic events. There is some evidence that the frequency and intensity of floods and landslides has increased in recent years, which can mainly be attributed to climate change (see, e.g., EEA 2004). It can be expected that this trend will continue in the future, and may even accelerate.

As a result, the extrapolation of natural hazards is not possible in the same way as for other soil threats, which occur continuously and wide-spread. Rather, to extrapolate the impacts of floods and landslides into the future, a modelling and forecasting approach is called for. This would need to be integrated with climate change models and forecasts of land use change. However, such an analysis is beyond the scope of this study.

The available cost figures cover the damage costs and the repair cost for damages to houses, infrastructure and other property. The indirect impacts of floods and landslides on the agricultural sector, industry and tourism are also considered in most cases, at least in qualitative terms or as rough estimates. There is some evidence that the off-site social costs caused by floods and landslides (e.g. through the disruption of transport routes and economic activities) constitute the biggest share of the total damage.

The linkage between floods and landslides and other types of soil degradation needs to be researched in greater detail. This applies both to the impact of floods and landslides on soil (e.g. through deposition of contaminated sediment), and to the influence of soil degradation on floods and landslides. While it is widely agreed that there are significant linkages between the two, these are rarely quantified in physical terms, let alone in monetary terms.

For these reasons, the estimations of the soil-related cost of floods presented above should only be regarded as a tentative first attempt to assess the dimension of the problem. These calculations should be supported with by further research in order to assess the contributions of different causative factors (including soil degradation in upstream areas) to the likelihood and intensity of flooding. This could be done, for example, through a limited number of case studies combining hydrological modelling with geophysical information on the state of soils and its impact on runoff dynamics. The same caveat applies to the cost of landslides: the data presented above can only provide some illustrations of the potential impacts, but does not allow for a systematic treatment on the European level.

A-1.3 Loss of Soil Biodiversity

Due to the limited data availability and the insufficient understanding of the causes and consequences of soil biodiversity loss, including the absence of empirical economic evidence, the discussion of the impacts of soil biodiversity loss will be confined to a qualitative description of possible impacts.

A-1.3.1 Situation

Although research on soil biodiversity has been carried out on the national level in European countries, no comprehensive and comparable data exist on the status of soil biodiversity at the European scale (WG on Research TG 3, 2004).

A-1.3.2 Method

Soil biodiversity and organic matter content in soil are directly related. Therefore, the impacts on soil and other environmental media, expected due to the decline of organic matter in soil, could be used to express impacts of the loss of soil biodiversity. The impacts could be grouped under the different cost categories (see Table 37):

Table 37: Impacts of soil biodiversity loss sorted into cost categories

Off-site costs	PC	▪ Soil biodiversity plays a central role in maintaining key soil functions and is directly responsible for soil formation, nutrient cycling, stabilisation of organo-mineral complexes, recycling of organic waste, infiltration rate and water holding capacity, and is thus an essential determinant of soil fertility. As a result, the loss of soil biodiversity impacts directly the fertility of soil and, hence the agricultural productivity.
		▪ In addition, the loss of soil biodiversity reduces both the resilience of soil to endure pressures, as well as soil bioremediation capacity and as a result could trigger other soil threats.
	MC	▪ Need of plowing, pesticides and fertilisation to replace or substitute lost soil functions.
On-site costs	SC	▪ Lost ecosystem services (i.e. bioremediation of chemicals, biocontrol of pests, waste recycling).
		▪ The loss of soil biodiversity leads to reduced capacity of soil to sequestrate C and could impact CO2 amount in the atmosphere.
	DC	▪ Cost of replacing lost ecosystem services (e.g. technical remediation vs. bioremediation).
	NC	▪ The loss of biodiversity in soil is a loss of certain species, as a result leading to changes in genetic resources present in soil, including moral and ethical consequences (WG on Research TG 3, 2004.
		▪ Impairment of landscape features in case of the loss of biodiversity in soil is not easy to notice; it has to become extreme to become visible.
		▪ Impact on patrimonial and bequest values from reduced soil resilience.

Soil biodiversity as an inherent value of soil is also affected by other forms of soil degradation, in particular soil erosion, decline in organic matter, contamination, acidification, salinisation and compaction. A particular link exists between the loss of soil biodiversity and soil erosion as well as decline in organic matter, as these threats reinforce each other.

A-1.3.3 Results

No data exist on costs related to the loss of soil biodiversity. So far, an economic value of the loss of soil biodiversity has not been assessed.

As the rich biodiversity in soil is an indicator of healthy soil, its loss leads to reduction of soil quality. This direct dependence is, however, difficult to evaluate economically because it is difficult to distinguish the loss of biodiversity in soil from other forms of soil degradation.

A-1.3.4 Interpretation

The loss of soil biodiversity is not fully understood from a natural science perspective. Consequently, the identification and quantification of its impacts is even more problematic, let alone the monetary valuation of the impacts.

Empirical data is very limited on the loss of biodiversity in soil, therefore, it is necessary to consider whether a modified form of benefits' transfer is possible between loss of biodiversity and the loss of organic matter in soil, e.g. whether extent and impacts of soil biodiversity loss could be approximated via organic matter content. For example, due to the direct dependency of organic matter content in soil and soil biodiversity, the costs claimed by decline of organic matter in soil could cover part of the damages of loss of soil biodiversity.

For the nearer future, however, it appears that the interdependencies between a decline of soil biodiversity and soil functions, and the impacts this has on soil uses, need to be investigated in more detail. Based on this, the economic valuation of the impacts would only be the third or fourth step of the analysis.

A-1.4 Soil Compaction

Due to the limited data availability and the lack of empirical economic evidence, the discussion of the economic impacts of soil compaction will be confined to a qualitative description of possible impacts, supported by quantitative evidence where available.

A-1.4.1 Situation

Compaction of soils reduces aeration of the soil, and consequently affects rooting density and rooting depth. It also has a negative impact on soil infiltration and on run-off potential. Almost all agricultural soils in developed countries are affected by soil compaction to a certain degree (WG on Research TG 1, 2004). EEA (1995) claims that soils sensitive to compaction are common in Belgium, north-western France, Germany, the Netherlands, Poland and Russia. In addition, Van Lynden (1995; 2000) in EEA (2003b) indicates that soil compaction is one of the main forms of soil degradation in Central and Eastern Europe. At the same time, the WG on Research, TG 1, 2004 recognises that it is almost impossible to avoid topsoil compactions entirely. However, through preventive measures, such as the design of agricultural machinery (wider tires, lower tyre pressure) and the appropriate timing of its use (depending on soil structure and humidity), compaction can at least be limited.

A-1.4.2 Method

Impacts on soil and other environmental media due to soil compaction could be grouped under the different cost categories (see Table 38):

Table 38: Impacts of soil compaction sorted into cost categories

On-site costs	PC	▪ Soil compaction impacts the physical and biological qualities of soil, hence reduces agricultural productivity.
		▪ Worsened plant growing conditions due to soil compaction result in higher vulnerability of crops to diseases (WG on Research TG 1, 2004.
	MC	▪ Cost of measures to loosen compacted soils and restore the physical soil structure.
		▪ Increased supply of nutrients and water to mitigate yield losses
Off-site costs	SC	▪ Reduced water infiltration into the soil means increased surface runoff, resulting in a higher risk of flooding, erosion and water pollution (e.g. transporting nutrients and agro-chemicals into water courses).
		▪ A poor aeration of soil due to soil compaction may cause a loss of soil nitrogen and emissions of greenhouse gases through denitrification in anaerobic sites.
	DC	▪ Indirect cost of measures to keep back rainwater runoff.
	NC	▪ Indirect impacts on landscape values and biodiversity.

Stoate et al. (2001) argue that the use of heavy machinery and frequent passes with cultivating equipment have caused soil compaction in many parts of Europe, increasing runoff at the soil surface and creating a soil pan within the soil. This soil pan has different impacts on plant growth:

- it inhibits drainage and causes waterlogging of crop plants on some soils;

- it creates a physical barrier for their roots, making them more susceptible to drought.

Soil compaction results in reduced water infiltration capacity and increases the volume of surface runoff (see e.g. Friedrich and Franken 2003). This accelerates other soil degradation forms, such as water erosion, floods and landslides. Soil compaction also reduces the quantity and quality of biochemical and microbiological activity in the soil. This indirectly affects organic matter development and soil biodiversity.

A-1.4.3 Results

The economic effects of soil compaction are covered only in occasional studies, or are not quantified at all. For example, the Literature Review (Volume I of this report, Table 13) indicates just one study that evaluates soil compaction in monetary values, namely Scrimgeour (1995) who presents the estimated willingness to pay for the protection against soil compaction in New Zealand to be equal to €$_{2003}$ 22 – 109/ha p.a. and lost output due to soil compaction in New Zealand to be equal to €$_{2003}$ 147 – 440/ha p.a. **(PC)**.

WG on Research, TG 1 (2004) points to the particularity that compaction and its impacts are more difficult to observe than the impacts of other soil threats. First, this is the case because soil compaction is a hidden form of soil degradation: soil compaction itself is not nearly as visible as e.g. erosion. Secondly, since soil compaction often occurs throughout a wider area, it may be difficult to find a comparable, non-compacted reference field. As this comparison is often not possible, the impact of soil compaction on productivity are difficult to observe. As a rough indication, the WG on Research, TG 1 cites evidence that compaction may reduce agricultural yields by 35% or more in extreme dry or wet periods.

The WG on Erosion, TG 5 (2004) acknowledges that while no precise data on the extent of soil compaction in Europe is available, about 32% of European subsoils are estimated to be highly vulnerable to subsoil compaction, and another 18% moderately vulnerable.

Combining the information on potential yield losses from compaction and the extent of compaction seems to suggest that yield losses could be as high as 10% of total agricultural yields. However, combining the two pieces of information is not easily possible: First, the indicative 35% figure cited by WG on Research, TG 1 is merely a rough indication, and not specify under which conditions it has been obtained. Secondly, the data cited by the WG on Erosion identifies the area that is vulnerable to compaction, rather than the area that is actually affected by it.

A-1.4.4 Interpretation

Soil compaction is widely distributed but tends to be most prevalent in agricultural areas and forest regions where heavy machinery is continuously used. In addition, farm structure should be considered, as compaction occurs more frequently on large farms. Contrary to the topsoil compaction, deep compaction of subsoil is accumulative, persistent and cannot easily be reversed (EEA,1995).

Economic information on the impacts of compaction is extremely scarce. This concerns both the direct, on-site impacts on agriculture and forestry and the off-site impacts, e.g. through reduced water retention and increased runoff. The few studies that have quantified economic impacts seem to suggest that the cost can be substantial, amounting to a considerable share of agricultural output, and running into hundreds of Euros. Experimental studies have concluded that measures to reduce soil compaction support the infiltration of rainwater, and can thereby support integrated flood protection strategies (Friedrich and Franken 2003). However, at this stage, this linkage cannot be quantified in economic terms.

A-1.5 Soil Sealing

Due to the limited data availability and the lack of empirical economic evidence, the discussion of the economic impacts of soil sealing will be confined to a qualitative description of possible impacts, supported by quantitative evidence where available.

A-1.5.1 Situation

EEA (2003b) finds that Belgium, Denmark and the Netherlands are the countries with the highest share of built-up area (between 16 per cent and 20 per cent of total land area). EEA-UNEP (2000) and EEA (2003b) indicates that in the Mediterranean regions, i.e. southern France, Italy, southern Spain and the Mediterranean islands, urbanisation has been increasing in the coastal zones, where tourism is the main driving force. In addition, pressure is increasing in some coastal zones of the Baltic Sea region, for example along the Baltic Sea coast of Germany, Latvia and Russia (Coalition Clean Baltic, 2002; EEA, 2003b). Baltic Environmental Forum (2001) and EEA (2003b) give an overview of the soil sealing situation in Central and Eastern Europe. The sources state that since 1990, the development of new infrastructures, the migration of rural populations to the cities and the development of new settlements lead to consequent increases in soil sealing in Central and Eastern Europe. Slovakia and the Czech Republic have the highest percentage of built-up area in Central and Eastern Europe (about 8 per cent of the total land area).

A-1.5.2 Method

Due to soil sealing, most of the natural soil functions are hampered, although not all of them are completely disrupted. In addition to these direct impacts on soil, soil sealing can also affect other environmental media. Both direct and indirect impacts due to soil sealing could be grouped under the different cost categories (see Table 39):

Table 39: Impacts of soil sealing sorted into cost categories

On-site costs	PC	▪ Opportunity cost of alternative uses of land.
	MC	▪ Cost of de-sealing measures.
Off-site costs	SC	▪ Soil sealing can have a major impact on water quality: runoff water from housing and traffic areas is normally unfiltered and may be contaminated with harmful chemicals.
		▪ Fragmentation of habitats and disruption of migration corridors for wildlife (EEA 2003b).
	DC	▪ Indirect cost of measures to keep back rainwater runoff.
	NC	▪ Impact on landscape and amenity values and on biodiversity.

A-1.5.3 Results

In the course of this analysis, no data could be retrieved where the costs related to soil sealing had been identified.

On the one hand, it is evident that soil sealing has direct and indirect impacts, as it goes along with a partial or complete loss of all soil functions in the sealed areas. On the other hand, soil sealing is an intentional process, unlike other types of soil degradation, which appear as unintended by-products of other activities. Therefore an analysis of the economic impacts of soil sealing has to pursue a different approach than other soil threats. As soil sealing occurs intentionally, the economic benefits derived from construction and soil sealing have to be taken into account and weighed against the ecological impacts, e.g. in environmental impact assessments for large infrastructure projects or other large buildings.

Other than having a direct impact on soil functions, soil sealing has also been identified as one contributor to the occurrence of floods, in that soil sealing changes water flow patterns and increases the runoff of water (PIK, 2000; EEA, 2003b). Following the August 2002 Elbe flood, the German Federal State of Saxony has introduced new legislation which limits soil sealing in upstream catchments, and which provides for the de-sealing of sealed areas as a compensatory measure for all new construction in catchment areas (see also A-1.2).

A-1.5.4 Interpretation

Soil sealing has the greatest impacts in urban and metropolitan areas, where large areas of the land are covered with buildings and infrastructure.

According to EEA (2003b), soil sealing in Western Europe is mainly the result of a steady increase in the number of households and average residential space per capita since 1980. This trend has accelerated since 1990 (EEA, 2001c; 2003b). At the same time, road infrastructure increased, adapting to increasing travelling distances (EEA, 2000a; 2003b). The demand for both new constructions and better transport infrastructures continues to rise. Soil sealing can be regarded as part of land use, and as the last step within the consumption of land for human use (WG on Research TG 5, 2004). According to the European Commission (2002), soil sealing is almost irreversible.

From an empirical perspective, the economic evidence on the impacts of soil degradation is too limited to provide an assessment of the impacts of soil sealing in economic terms. From a methodological perspective, however, it is clear that soil sealing differs from other threats treated in this study, because soil sealing is inextricably linked to construction and thus occurs intentionally. An economic discussion of the impacts of soil sealing would therefore also have to reflect the (expected) economic benefits and weigh these against the cost.

Consequently, the information on the economic impacts of soil sealing could be integrated into a Cost-Benefit-Analysis or an environmental impact assessment, e.g. for a development plan or for large infrastructure projects.[34]

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[34] A further application for the economic valuation of soil sealing could arise from the introduction of tradable permits for construction ground. Such a system has recently been proposed for the German *Bundesland* of Baden-Württemberg (NBBW 2004). If implemented, it would allow the creation of a market price for soil sealing that would reflect its scarcity, as defined through political objectives.

Annex 2: Methodology and Data Needs to Assess the Cost of Contamination

This annex aims at identifying the information that would be needed to assess the different component of the costs listed above. As illustrated by the Table 8, each and every type of cost is not likely to be encountered in all contaminated sites. Whereas a complete assessment may be required for highly contaminated sites (level 3 of the EEA classification system, a simplified evaluation may be carried out for others (level 1 and 2 of the EEA system). The table below gives a brief definition of the EEA's contaminated sites level.

Table 40: Impact levels applied to contaminated sites (EEA, 2002)

Level	Brief definition
Level 0	No impacts; no use restrictions (mostly applied at remediated sites)
Level 1	Minor impacts (tolerable contamination); no use restrictions
Level 2	No significant impacts under current use of environmental media, restricted use only
Level 3	Significant impacts, action needed

A-1.1 On-site private costs (PC & MC)

A-1.5.5 Monitoring measures & impact assessment studies:

Most of the sites declared as contaminated are under strict monitoring in EU countries under different legal frameworks (e.g. IPPC directive, landfill directive). The costs of monitoring and impact assessment can be divided up into three main components related to the impact of the contamination on (i) human health (population survey); (ii) on groundwater; and on surface water (including sediments) and (iii) on other ecosystem impact.

The cost of *public health monitoring* is assumed to be proportional to the exposed population (as it is plausible to assume that the total impact will be higher in the Ruhr area than in the French Massif Central). Assuming that all contaminated sites can be classified into two categories (with or without health risk), an aggregate annual cost can be estimated as follows:

$$C_{mh} = N_{rh}\ c_{mh}$$

where : N_{rh} is the number of sites presenting a risk for human health;
c_{mh} is an average cost of human health impact monitoring per site (which can be calculated using a few case studies in different contexts)

The cost of *water monitoring* (surface and groundwater) can be assessed with similar assumptions: all sites are classified into two categories (with and without risk of groundwater contamination). An aggregate estimate of the annual cost is assessed as follows:

$$C_{mw} = N_{rw}\ c_{mw}$$

where : N_{rw} is the number of sites presenting a risk of water contamination;

c_m is an average cost of water quality monitoring (which could be calculated based on a few case studies with different surface and groundwater resources).

The cost of *monitoring of ecosystem impacts* can hardly be assessed at an aggregated level as it is likely to be highly site specific. This cost is likely to represent a minor component of the total monitoring cost and can therefore be neglected.

A-1.5.6 Decontamination of the site (clean-up):

Assuming that all sites which have been decontaminated over a period of T years (for instance the last 20 years) can be classified into 3 – 4 classes with regard to the extent of soil contamination, a very crude estimate of the aggregate decontamination annual cost (C_d) could theoretically be estimated as follows:

$$C_d = \frac{\sum_i N_i \ c_{d,i}}{T}$$

where : *i* is the class of site (1 to 3 or 4)
N_i is the number of sites which have been decontaminated during the last T years;
$c_{d,i}$ is the average decontamination cost of the sites of class i

The estimation of the average annual decontamination cost C_{di} is however likely to be very difficult, and would have to rely on a number of strong assumptions. Indeed, the cost of clean-up of a specific contaminated site not only depends on its size (in ha) and on the depth of the soil to be excavated and treated, but also on other parameters such as the number of contaminants to be removed (frequent cases of multiple contamination), the remediation target levels and the clean-up technology used.

A-1.5.7 Redevelopment of the sites

Often, clean up activities do not lead to a full restoration of former contaminated sites to pristine conditions. This approach, established 30 years ago, is no longer applied. Current approaches focus on sustainable solutions, which will restore the usability and economic value of the land (CLARINET, 2002). These solutions can be characterised by three elements, the two first ones describe the environmental goals including spatial planning aspects, and a third one describing the way these goals ought to be achieved:

(i) *Fitness for use*: This aims at reducing human health risks and ecological risks as necessary to permit the safe (re)use of the land. It is focussed on quality requirements of the land for uses and functions.

(ii) *Protection of the environment*: this can be achieved by preventing the dispersion of pollutants to the surroundings. This is not an issue that only depends on the uses and functions of the land itself, but may also be dependent on the uses and functions of the surrounding land. Moreover the way the “dispersion risk” is addressed may be different from risks under the "fitness for use" heading. For example hindering further spreading of pollution by surface water and groundwater may be seen as a form of risk reduction, but the interpretation of risk in this case is more than mere toxicological risks.

(iii) *Reduction of aftercare*: If a solution is chosen which leaves immobile or inaccessible contaminants in the soil there is a need for aftercare. Monitoring and control may be necessary. Sustainable solutions minimise the burden of aftercare. Endless pump and treat solutions or containment walls that require control and maintenance forever may achieve fitness for use and prevent pollution of surrounding areas, but may be less desirable in view of the amount of aftercare required and the associated costs.

In this way, the target contamination level after treatment is such that the site does not present significant health and/or environmental risks. It can however not be used for certain activities, residential development, etc. and it is often reconverted into an area for industrial or commercial activities. The cost of redevelopment can be significant.

In the absence of existing data, one can assume that a certain percentage of all large decontaminated sites are redeveloped (i.e. only one or two of the site classes defined above). This percentage could be assessed at the regional or national levels by experts from public agencies – as those are frequently involved in financing the redevelopment (for instance ADEME in France). An average cost for redevelopment, based on a limited number of case studies, could also be used for extrapolation.

A-1.5.8 Acquisition and protection of contaminated land:

Where the contaminated soil does not represent a serious threat to the environment (air, water) or to human health, and if the costs of decontamination are prohibitive regarding the expected benefits, protection of the contaminated site may be preferred to decontamination. The contaminated site is enclosed, sometimes turned into forest and the access is restricted. The purchase of land from third parties represents the main component of the cost. The cost of protection is directly proportional to the size of the site; it also depends on the location of the site as the price of land may vary from 1 to 20 between remote rural areas and dense urban areas.

An estimate of the cost of protection could be assessed as follows:

$$C_p = \left(\sum_{i=rural} N_i\, c_{p,i} + \sum_{j=urban} N_j\, c_{p,j} \right) / T$$

where : N_i is the number of sites which have been protected in rural areas
N_j is the number of sites which have been protected in urban areas
$C_{p,i}$ is the average protection cost for sites in rural areas
$C_{p,j}$ is the average protection cost in urban areas
T is the reference period (20 years) over which the estimate is calculated.

This estimate can only be done on the basis of a census of the sites where protection has been implemented, and on the basis of additional case studies conducted in different contexts and regions, in order to provide estimates of the average cost of protection.

A-1.5.9 Decontamination of sediment (in surface waters)

Erosion of contaminated soil can lead to the contamination of sediments in rivers, canals and other surface water bodies. Using the treatment cost approach, one can consider that the damage cost of sediment contamination is equal to the cost of removing the contaminant from the sediments (sediment dredging, decontaminating and storing). This cost depends on the volume of contaminated sediments, which in turn depends on highly site specific factors: distance of the contaminated soils from the surface water bodies at risk, protection measures implemented on site, duration of the contamination, etc. Assessing an aggregate estimate of this cost would at least require knowing the number of sites likely to have contaminated water bodies – information which does not exists today.

A-1.6 Social costs

A-1.6.1 Human health impact

The cost of human health impact can be assessed using the cost of illness method, which assumes that the economic loss is equal to the cost of illness treatment and the foregone income of workers affected. For a given site and given contaminant causing a specific illness, the cost of human health impact is equal to: $(r \cdot P_t \cdot c_h) + (r \cdot P_a \cdot w \cdot d)$

Where P_t is the total population potentially concerned;
P_a is the working population;
r is the % of population actually affected by contamination;
c_{mt} is the cost of medical treatment per affected person, which depends on the type of contaminant;
d is the average number of days of inaptitude for work per affected worker (depends on the type of illness);
w is the average daily wage.

A crude estimate of an aggregate cost can be made with the following assumptions: (i) only a restricted list of contaminants have a significant impact on health; and (ii) population is significantly exposed in a limited number of sites. Assuming that the contaminants are classified into X classes (e.g. radioactive nuclide, heavy metals, pesticides, etc) the extrapolation of the cost can be carried out as follows:

$$C_{health} = \sum_{i=1}^{X} (r P_t c_{i,h} + r P_a w d_i)$$ with the same notations as above.

Soil contamination may also generate significant fear and anxiety among the population– especially in the case where it is proven that the polluting activity has had a significant impact on human health in the past. The fear and anxiety itself could also be seen as part of the damage (stress-related symptoms leading to loss of subjective well-being etc.) – as they may have a considerable affect on well-being and productivity.

A-1.6.2 Agricultural impact

Contamination of agricultural land by a limited number of toxic substances result in either the impossibility to grow crops or in reduced quality or yield of the crops. Overall, this translates into a loss of income per hectare. This loss depends on the type of contaminant, on the extent of the contamination but also on the intrinsic quality of the land and on the crops grown by the farmers. The maximal value of the loss is equal to the price of agricultural land, but it can be lower if the contamination only reduces the yield or the quality of the product, without making crop cultivation impossible.

For the sake of the extrapolation, we assume that :

(i) only a limited number of contaminants have a negative impact on agricultural use of the soil (X classes);
(ii) the loss of income due to the contamination translates into a loss of value of the land. This loss of value is noted $\alpha_i V_a$, with $0<\alpha<1$ and V_a the market value of agricultural land;
(iii) each type of contaminant can be related to a type of polluting activity for which it is possible to assess an average area contaminated S_i (for instance, the pollution by a Pb smelter extends on 3 to 4 kilometres around the plant)
(iv) the number of sites contaminated by the X classes of substances can be estimated.

With these assumptions, the aggregate cost of contamination for agriculture is equal to:

$$C_{agri} = \sum_{i=1}^{X} N_i\, S_i\, \alpha\, V_a$$

A-1.6.3 Loss of value of urban land (land use restrictions)

Public agencies may have to impose land use restriction on contaminated land. This may generate huge losses for land owners, in particular in areas where construction would have been allowed by the local land use master plan. An approach similar to the one presented above (agricultural land) can be implemented, assuming that:

(i) only a limited number of contaminants showing a risk for human health impose that land use restriction be implemented (X classes);

(ii) areas which are downgraded from "building land" to "non building land" lose approximately 90% of their economic value, noted V_u (an average value can be used as a proxy).

(iii) each type of contaminant can be related to a type of polluting activity for which it is possible to assess an average area contaminated S_i

(iv) the number of sites contaminated by the X classes of substances can be estimated (N_i).

$$C_{urban} = 0{,}9.\sum_{i=1}^{X} N_i\, S_i\, V_u$$

A-1.6.4 Loss of value of real estate

= Soil contamination can results in a decrease of the market price of real estates, the local population preferring not to live in an area perceived as exposed to contamination. As shown in the MetalEurop case study in Volume II of this report, this indirect cost can be significant for households owning a house in the contaminated area.

A possible approach for assessing this cost would consist in using the results of all hedonic pricing studies (which assess the loss of value due to the presence of contaminated site) and to transfer the results found in these studies to other sites (benefit transfer approach). We would consider, for instance, that for a given range of pollution activities, all houses located within a certain distance d from the site lose a percentage *k* of their market value. This distance depends on the type of contaminant and the source of pollution (definition of X classes of contaminant).

For extrapolation, at least 3 classes of urban density may be created in order to differentiate sites located in dense urban, lose urban or rural areas. For each class *j*, and for each of the X classes of contaminant, an average loss of real estate value can be assessed (noted Ci,j below). The extrapolation is then carried out, based on an estimate of the number of sites, as follows :

$$C_{estate} = \sum_{i=1}^{X} \sum_{j} N_{i,j} C_{i,j}$$

A-1.6.5 Damage cost for drinking water utilities (shut down of wells)

Contaminants leached by rainfall sometimes reach the aquifer, possibly leading to major contamination of drinking water wells. This is only likely to occur for a limited number of contaminants and for sites underlain by groundwater. Extrapolating this cost cannot be made without having access to a list of drinking water wells which have been shut down for this reason.

A-1.6.6 Damage cost on ecosystems and related users (fishing, etc)

Contamination may have an indirect impact on the quality of ecosystems, which can be affected by air contamination (dust), water pollution, etc. Wildlife habitat, animal and vegetal population can be affected. This may create an indirect cost for people deriving benefits from the environmental quality of these ecosystems (sport fishing, walkers, bird watchers, etc.). However, while it could be possible to assess the value of the ecosystems themselves (with the benefit transfer method for instance), either for direct human uses or indirectly as a source of ecosystem services, it will be very difficult to identify how this value is affected by contamination from a specific source. The total cost of ecosystem damage is therefore not assessed as part of this empirical evaluation.

A-1.7 *Defensive cost*

A-1.7.1 Measures to mitigate impact on groundwater (pumping)

The cost of groundwater protection measures, aiming at avoiding pollution plume extension can be assessed making similar assumptions as for the cost of monitoring above. All sites can be classified into two categories (with and without risk of groundwater contamination). An aggregate estimate of the cost of pumping is assessed as follows:

$$C_{mw} = (N_{rw}\ c_p) / T$$

where : N_{rw} is the number of sites presenting a risk of water contamination;
cp is an average cost of water protection measures , for instance pumping (which could be calculated through a limited number of additional case studies with different surface and groundwater resources).

A-1.7.2 Other defensive cost

Measures aiming at confining the contaminated sites (covering with geo-membranes for instance) can generate significant costs. Such measures are however not widespread – it is therefore not a priority to assess them. The same remark can apply to the cost of measures aiming at mitigating the impact on surface water.

A-1.8 Summary

Assessing this cost at an aggregate level (regional, national or European) therefore seems to be a high priority task to inform policy makers and support strategic decisions. However the information available today is clearly not sufficient to conduct this economic assessment. The preceding paragraphs illustrate that two types of information have to be acquired:

- First, undertaking a census of contaminated sites seems to be a high priority task. The census would enable to characterise all sites with regards to: the number and the type of contaminants; the extent of the contamination (area and depth of the soil); the population exposed; water and other ecological resources at risk of contamination; etc (see table below). The information could then be used to build a typology of sites (through clustering for instance) which could be used as a basis for assessing costs at an aggregate level.

- Second, a series of case studies have to be conducted, using a sample representative of EU situations, in order to assess average cost values (with upper and lower bound values) that could also be used for an extrapolation (see ables bellow). Because such values do not exist today, aggregate cost values are assessed based on assumptions.

Table 41: List of minimum information needed for all sites

Variable	Description
Type of contaminant(s)	List
Location	Dense urban / Urban / Rural
Size of the site	With classes from 1 to 10
Size of the contaminated area	Idem
Population (total and active) concerned by the risk of contamination	With classes from 1 to 10
Risk for human health	High / moderate / low
Risk for water resources (groundwater and surface water	High / moderate / low
Decontamination measures implemented – Date	Yes / No
Redevelopment of the site	Yes / No
Measures to protect the site implemented, acquisition of the land (compensation) – Date	Yes / No

Table 42: Average costs to be assessed through a limited number of case studies

Average cost value to be assessed	Number of values	Notation above
Average cost of human health impact monitoring	1 average value	$C_{m,h}$
Average cost of water resources monitoring	1 average value	$C_{m,w}$
Average decontamination cost	X average values (for each class of contaminated site - class being defined with regards to the nature of the contaminant and the extent f the contamination)	$C_{d,i}$
Average cost of protection measures	2 average values, one for urban area, one for rural area	$C_{p,i}$ and $C_{p,j}$
Average cost of illness treatment	X value (one for each major hazardous contaminant)	$C_{i,h}$
Percentage of population affected (illness) (rural, urban)	1 value per contaminant and for 3 levels of population density	r_i
Average loss of loss of property of real estates in the vicinity of a contaminated site	3 values – one for each class of urban density	$C_{i,j}$
Average cost of groundwater protection measures	1 value	C_p