European Commission

DG Environment

Economic Assessment of Groundwater Protection: A Survey of the Literature

Final Report

A project financed by the European Commission

ENV.A.1/2002/0019

Benjamin Görlach

Eduard Interwies

April 2003

Keywords: groundwater, economic instruments, protection, remediation, valuation, ecosystem benefits, cost-benefit analysis

This report will be quoted as follows:

Görlach, B. and E. Interwies (2003): Economic Assessment of Groundwater Protection: A survey of the Literature. Berlin: Ecologic

Economic Assessment of Groundwater Protection:

A Survey of the Literature

Table of Contents

1 Introduction

1.1 Scope and Content of the Study

This study was written by Ecologic, the Institute for International and European Environmental Policy, as part of the project "Economic Assessment of Groundwater Protection". This project was commissioned to Ecologic and the French Geological Survey BRGM by the DG Environment of the European Commission, in support of the development of the future Groundwater Directive.

The main objective of the literature review has been to report results from case studies that have assessed the costs and the benefits of groundwater protection and remediation, either as a qualitative description or, where possible, in monetary terms. In assessing the costs and benefits of groundwater protection, this paper will take account of the focus of the Water Framework Directive, which is primarily concerned with the qualitative aspects of groundwater protection. Quantitative aspects will only be considered in so far as they have a connection to qualitative problems.

The report builds on evidence from scientific research papers, literature by international organisations, and publications by government agencies, consultants or other stakeholders. In order to identify potential unpublished reports, a "core contact group" of experts in the field of groundwater economics has been contacted with a specifically designed questionnaire.[1]

1.2 Outline of the Study

This study is structured around three main parts. The first part, comprising the chapters 2 and 3, gives an overview of the most relevant issues in the relationship between groundwater and economics.

- Chapter 2 provides a short introduction to some main concepts of **environmental economics**, and the peculiarities encountered when applying these cases to groundwater.
- Chapter 3 surveys the **main factors** that influence groundwater pollution from an economic perspective. It first considers different *sectors* and their contribution to groundwater pollution, and then surveys the most relevant *types* of pollution. Finally, it highlights some examples of particular *hydrogeological or biological situations* that must be considered in an economic assessment of groundwater pollution.

The second part of this study, consisting of the chapters 4 to 6, offers an overview of the empirical evidence from economic studies that have highlighted different aspects of the economic analysis of groundwater protection. As very few of the reviewed studies comprise both the costs and the benefits of groundwater protection measures in a comprehensive way, instruments, costs and benefits will first be discussed in separate chapters.

- Chapter 4 discusses studies that have compared the usefulness and efficiency of different **instruments** and measures to protect groundwater from pollution, or to restore polluted groundwater bodies to their original state where this is possible. It comprises economic instruments, such as taxes or cooperative agreements, as well as classical regulations.
- Chapter 5 presents evidence of the **costs** of implementing some of these measures.

[1] The authors would especially like to thank the following experts for their contributions and comments: Rob Curry of the UK Environment Agency; Victor Dries of the Flemish Public Waste Management Coorporation (OVAM); Ingo Heinz of the University of Dortmund; Lena Ojala of the Swedish Geological Survey (SGU); Tore Söderqvist of the Beijer Institute for Ecological Economics, Stockholm; and Onno van Sandick of the Dutch Ministry of Public Housing, Spatial Planning and the Environment (VROM).

In addition, we would like to thank Jean-Daniel Rinaudo of the French Geological Survey BRGM, as well as Tanja Dräger, Nicole Kranz, Britta Pielen and Maic Verbücheln of Ecologic for thoughtful comments and contributions.

- Chapter 6 deals with the **benefits** of groundwater protection. It discusses the different approaches used to place a value on a non-tradable good such as good groundwater quality, and surveys the results from different empirical studies. Benefits of groundwater protection are typically estimated in a reverse approach, by calculating the costs that groundwater pollution imposes on the different users, under the assumption that these costs could be saved if groundwater quality were improved. In this context, the costs of pollution are not restricted to direct economic costs only: they also include the contribution of groundwater to dependent ecosystems and surface water bodies.

Finally, the third part of this study assesses to what extent the costs and benefit estimates presented in the previous chapters can be combined, and what can be derived from them.

- Chapter 7 discusses different approaches of combining costs and benefits, ranging from a full economic cost-benefit-analysis to simpler tools, which permit the incorporation of economic effects without the need for a full economic assessment. The impact of risk, uncertainty and data limitations will also be discussed in this context.
- Chapter 8 offers some conclusions from the analysis and possible implications from the results of this study.

2 Environmental Economics and Groundwater

This chapter will briefly explain some of main concepts from the theory of environmental economics, and explain how they are related to the economic analysis of groundwater protection. Environmental economics is a relatively young discipline. Its principal interests are to explain

- how the forces of the market may lead to environmental damage,
- which instruments should be used to correct this, and
- which level of environmental protection will be optimal for society as a whole.

One of the central assumptions in standard economics is that, under normal conditions, the free play of market forces will achieve the socially optimal result: the 'invisible hand' of the market will guarantee that while all agents only pursue their own interests, they achieve a result that is also optimal from the point of view of society as a whole. In some cases, however, the market will fail to achieve the socially optimal outcome. A targeted government intervention can then increase social welfare. Two categories of such market failures are relevant in the case of groundwater: first, the case of externalities, and second, the public good aspect of groundwater.

2.1 Externalities and the theory of optimal pollution

In welfare economics, an externality is present when an economic activity has side-effects on a third party, which do not enter the cost-benefit considerations of the decision maker. In plain language, externalities arise when a producer (or consumer) inflicts damage on others, for which he does not compensate them.[2] As a result, the social costs of this activity (i.e. the costs borne by society *and* producers) are higher than the private costs (i.e. the cost faced by the producers only). This is relevant, because only the equality of *social* costs and private benefits ensures a socially optimal outcome, as will become clear in the following example.

Box 2.1: Example of a negative externality in the case of groundwater

A typical example of a negative production externality in the field of groundwater would be nitrate pollution from a farm, which affects households and firms drawing their groundwater supply from the same aquifer. If the farm faces no regulation whatsoever, the farmer will not consider nitrate pollution in deciding the amount of fertiliser used - the level of output from the farm will then only be determined by 'internal' considerations such as the available land, and the cost of seeds, machinery, labour, and fertiliser. However, the crucial point is that the farmer will only consider the *private* costs of fertiliser, i.e. the price that he has to pay for it. He has no incentive to take into account the *social* costs as well, which include negative effects on third parties.

Through nitrate leaching from the fields, the farm imposes costs on adjacent households and firms. Whether the neighbours actually suffer a direct financial loss because of the pollution - e.g. a brewery using the groundwater, which now has to install purification devices - or whether their loss consists in a reduced quality of life, is irrelevant in this context. The crucial point is that the neighbours of the polluting farm would benefit if the farmer reduced his nitrate applications. Society as a whole (i.e. the polluting farm and its neighbours taken together) would also reach a net benefit, as long as the cost of reducing

---

2 Externalities need not be negative, however: there are also examples of positive externalities, where third parties enjoy benefits from the economic activity of others, for which they do not have to pay. An example would be someone who lives next to a concert venue, and gets to hear the music without paying for a ticket.

For an introduction to the theory of externalities, see e.g. Baumol and Oates (1988), Tietenberg (1992) or Hanley, Shogren, and White (1997).

nitrate pollution is lower than the benefit from it. Note that it does not matter whether pollution is reduced because of an improved irrigation management system, or because the farmer simply reduces nitrogen application across the board, and thereby also lessens his output. In the former case, the cost of abatement equals the cost for installing the irrigation management; in the latter case, the cost consists of foregone harvest.

At the same time, this system will not lead to the elimination of all pollution. Rather, the optimal pollution level is reached at the point where the benefits from reducing pollution by one additional unit are just equal to the cost of this reduction - or, in economic terms: where the marginal net social benefit equals the marginal abatement cost.[3] What level of pollution is considered optimal depends on a variety of factors: this paper presents a range of different ways by which the costs and benefits of reduced pollution levels in groundwater can be assessed, and not all of these ways lead to the same results. One standard assumption in economics is, however, that the initial pollution reductions are cheap when there is little regulation, and grow more expensive subsequently because the cheapest abatement options are exhausted first. On the other hand, it is commonly assumed that the benefits of pollution reductions are greatest in the initial stages, whereas the benefits of moving from clean to very clean water are lower.

2.2 Coasean contract solutions and Pigouvian taxes

In the presence of externalities, the market will not allocate resources optimally: if a good has negative external effects, too much of it will be produced compared to the social optimum; if production or consumption leads to positive externalities, too little of it will be provided from the point of view of society as a whole. Therefore an intervention that internalises the externality can improve social welfare and correct the market failure to a degree. In environmental economics, there are two classical ways how external effects can be incorporated. First, the affected parties may enter into direct negotiations and find a compensation scheme where all parties maximise their benefits; these solutions are known as Coasean or contract solutions.[4] Secondly, the state may levy a tax from polluters for their polluting action, and thereby correct their incentives; these instruments are referred to as Pigouvian taxes.[5] The instruments and the underlying logic will now be explained briefly.

Coasean solutions consist of cooperative agreements between polluters and the victims of pollution. Which form the agreement will take depends on the distribution of property rights: if the polluter has the right to pollute, then the victim will have to compensate him for cutting back on pollution; if the victim has the right to an unpolluted environment, then the polluter has to compensate the victim for accepting some pollution. The underlying idea is that in a situation with high pollution, the victim values a pollution reduction very highly, while the polluter has enough cheap options to cut back on pollution. Alternatively, if there is little or no pollution, the victim will not care too much about some additional pollution, while the polluter can gain from a sizeable cost reduction if environmental regulations are reduced by a certain amount. Therefore, there is considerable scope for gains from such deals. On the other hand, such solutions depend on a number of preconditions: the legal right to pollute, or to live in an unpolluted environment, has to be clearly allocated; the number of actors involved must be small – the more people are involved, the less likely they are to agree; and finally, all actors have to have a clear idea of the economic interests at stake.

3 In most cases, the optimal pollution level will be a hypothetical benchmark rather than a realistic policy objective: in practice, the so-called transaction costs limit the efficiency of the described process. Transaction costs comprise i.a. the costs of negotiating, implementing and monitoring policy measures.

4 named after Ronald H. Coase, who first suggested negotiated contract solutions as an efficient way of internalising externalities (Coase 1960), and showed that they would lead to the same social optimum as taxes.

5 named after Arthur Cecil Pigou, who first introduced the idea of social welfare into economic theory put forward the idea of internalising external effects and thereby (Pigou 1920).

One typical example of such a Coasean solution in the context of groundwater are the cooperative agreements (cf. chapter 4.1.3). In these cases, farmers have agreed to change their land use practices in groundwater catchment areas, i.e. by using less or different nitrate fertilisers. In turn, they receive some compensation from the water suppliers, who can save on water treatment and purification. In principle, such deals are beneficial for both sides as long as the reduced treatment costs outweigh the cost of reducing fertiliser application.

The second group of instruments is that of *Pigouvian taxes*. In these cases, the state decides which party has to compensate the other, and levies taxes accordingly. The idea is simple: since the unregulated polluter does not have to pay the full cost for his action[6], and therefore produces (or consumes) too much of the good in question, the state imposes a tax that internalises the external costs. Thereby the incentives for the producer (consumer) are changed, so that less of the polluting good is produced (consumed) (cf. e.g. Baumol and Oates 1988). This idea underlies all kinds of eco-taxes, whether they are based on the energy consumption, or on the emission of harmful substances, or on the consumption of raw materials; in all cases there is a perception that market prices fail to mirror the society's valuation of the environmental asset under threat. In the case of groundwater protection, one example of a Pigouvian tax would be a levy on pesticides on nitrates that is raised in order to reduce their use.[7]

One peculiarity about taxes is that only the price of the polluting action is changed; whereas the amount of pollution only changes indirectly as a consequence to the price change. Therefore, if the policymaker is at error about the polluters' response to taxation (the *price elasticity* of demand / supply), it is possible that taxation does not lead to the desired reduction in polluting activities. Therefore, in cases where there is much uncertainty about how consumers will respond to a price change, or where additional pollution could lead to severe damage, it is preferable to regulate the amount of pollution directly, e.g. by imposing a standard.[8]

2.3 Groundwater as a public good

Next to the problem of external effects, there is a second reason why unregulated markets may fail to ensure groundwater protection at the socially optimal level: groundwater has some characteristics of a public good. The distinction between public and private goods is common in welfare economics.[9] The underlying idea is that a pure private good has two central features: there is *rivalry* in consumption, i.e. if one person uses it, its value for someone else declines; and there is *excludability* from consumption, which means that it is possible to restrict access by others. A pure public good, by contrast, has neither of these characteristics. The textbook example of a public good is a lighthouse: neither is it possible to restrict its services to some ships only, nor is there any disutility for one captain if another ship is guided by the same lighthouse.

It is commonly accepted that markets only work efficiently for private goods. In the case of public goods, few people would be prepared to pay a price that equals their benefits from using the good: if they cannot be excluded from using the good, there is an overwhelming incentive to free-ride (to use it without paying), and since the same good can serve any number of users, it is not possible to assess the marginal cost of providing it. For a public good, this bears two negative implications: the good will be underprovided, and it will be over-used. The fact that the free market would produce too little of a public good is the reason why they are usually provided through the government; examples of this are education and national

---

6 This applies in the case of negative externalities – in the case of positive externalities the logic is the same, only with the opposite signs.

7 Given the fact that applications of nitrate and pesticides take place in the agricultural sector, it is also possible that subsidies may be designed accordingly, rather than directly taxing fertiliser or pesticide use. In its effect, the two are comparable.

8 This idea has been formally elaborated by Weitzmann (1974) and is also known as the Weitzmann Theorem.

9 For an introduction to public good theory, refer to Stiglitz (2000). A classical article in this context is Lindahl (1919).

defence, but also stable financial and legal system. The over-use problem is also known as the "tragedy of the commons" (Hardin 1968): although it would be rational from the point of view of society to restrict individual use, the individual rationality dictates to use the public good beyond the social optimum.

Clean groundwater, in its capacities to provide drinking water and to absorb emissions, displays some characteristics of a public good.[10] On the other hand, there is also rivalry to a certain degree – if the groundwater body is used to absorb emissions, at some stage the water from it cannot be used as drinking water any longer; and if too many users abstract water from an aquifer, it will be exploited above its recovery rate – leading to falling water tables, and for small aquifers to their depletion. To what extent there is excludability, i.e. whether users can be excluded from using the groundwater body, is largely a legal and technical question. However, especially in the case of diffuse pollution, it is impossible to prevent or monitor all pollution, so there is at least partial non-excludability from consumption (i.e. the possibility to pollute). Economically speaking, groundwater can therefore be seen as a *common pool resource*, or as an impure public good (Tietenberg 1992). This characterisation also has implications for the role of governments and markets in protecting groundwater. Thus, the lack of physical excludability means that the government must act as steward over the resource, e.g. by preventing pollution through legal means. Also, the partial absence of rivalry means that the market will not yield prices for groundwater that are socially optimal; governments can therefore raise prices either indirectly, by restricting access to groundwater, or directly through taxation.

2.4 Conceptual Difficulties in Dealing with Groundwater

The instruments and approaches described above are largely taken from the standard toolbox of environmental economics. Unfortunately, in the case of groundwater, there are a number of factors that complicate the application of economic instruments straight out of the textbook: a report by the Environment Agency of England and Wales describes this as the "complexity of dealing with a moving, changing contaminant in a heterogeneous, dynamic medium, with uncertain data, in changing and imperfect market conditions" (Environment Agency 1999, p. 5). The following problems need to be considered:

- Groundwater contamination is subject to considerable *time-lags*: some contaminants travel for decades before they even reach the aquifer and cause pollution of groundwater; this makes it particularly difficult to monitor the effectiveness of protection measures. In addition, these time lags are *variable*: they themselves are influenced by a range of other factors, such as soil type, saturation, or precipitation. Once contaminants reach the groundwater body, they continue to spread, albeit at a slow pace. How quick contaminants reach the groundwater, how fast they spread and how long they remain there depends mainly on the kind of pollution.
- Moreover, the impact that contaminant release has depends on a range of factors: these include the *hydrogeolocial conditions* of the site, such as the thickness and soil type of the topsoil layers, the depth and volume of the aquifer, and its connection to surface water bodies. Other factors to be considered are *meteorological conditions*, such as the amount and frequency of rainfall that leads to leaching of contaminants from soil to groundwater.
- At the same time, the impact that groundwater contamination has will also depend on *groundwater uses*, such as the present and future groundwater abstractions for irrigation, drinking water or industrial uses, as well as the vulnerability of groundwater-dependent ecosystems. Unfortunately, many of the linkages between groundwater, surface water, and dependent ecosystems are poorly understood.

---

[10] Technically speaking, not the groundwater itself is the public good, but its *unpolluted state*. The difference is that the government does not have to provide the groundwater itself - its protection is what has to be safeguarded.

- One further peculiarity of groundwater is that damage makes itself felt for a long period, but is very difficult or impossible to correct: in most cases, pollution can at best be contained within a certain area; however a cleanup of polluted groundwater is usually not possible. The *irreversibility* of groundwater protection increases the cost of misjudgements when determining groundwater protection levels (cf. chapter 7.2)
- Finally, concerning the benefits of groundwater protection, a special property of groundwater is its invisibility to the general public. In contrast to the pollution of other natural resources, groundwater pollution can only be communicated through scientific terminology, e.g. by reporting pollutant concentrations. Consequently, groundwater protection is mainly viewed in relation to the protection of drinking water, whereas other functions are much less researched. This applies in particular to groundwater-dependent ecosystems and surface water bodies: these involve a much larger fraction of total groundwater than human uses do, however much less is known about the effects of groundwater contamination in these cases, and the economic costs associated with it (cf. chapter 6.4)

This host of caveats, limitations and influences means that an assessment of groundwater pollution and protection will be largely determined by local characteristics, and will have to be done in a site-specific way. In this context, the presented instruments from environmental economics can then be applied in a useful manner. The difficulties listed here will be discussed in more depth in later chapters, along with their implications for the application of economic instruments and procedures.

3 Groundwater Pollution

The focus of this chapter is to give a survey of the main issues that can be distinguished in the field of groundwater pollution. The most frequently used subdivision is to structure the issue by looking at particular economic sectors and the pollution problems they face. A second approach, followed by some studies, is to consider a particular pollutant, or a type of pollution (point-source or diffuse pollution). Finally, it is also instructive to consider the different regional, geological and biological conditions in which groundwater pollution occurs. To offer a full classification of these conditions would far exceed the limits of this survey; therefore, the discussion will be restricted to some of the main cases. These include the effects of groundwater pollution on dependent wetlands, pollution of shallow aquifers, pollution and degradation in the case of coastal aquifers, and pollution of catchment areas where aquifers are recharged.

Obviously, the different categorisations of groundwater pollution issues overlap to a degree – nitrate and pesticide pollution are typical problems of the agricultural sector, whereas point source pollution is typically connected with industrial pollution, such as industrial waste, oil spills or leakage from underground storage tanks. Most of the studies considered here regard groundwater mainly as a reservoir of clean water that can be put to different economic uses (irrigation, drinking water etc.), and consider the effect that pollution has on these uses.

However, as Bergstrom et al. (1996) underline, this is only one of two central functions that groundwater has: its other function is to support surface water flows, such as springs and wells, as well as groundwater-dependent ecosystems. Surface water flows, in turn, support a range of services that are both economically and ecologically relevant (such as fisheries, water supply for agriculture and forestry, support of biodiversity, amenity and recreation, etc.); the same applies for groundwater-dependent ecosystems. These benefits of groundwater protection that are permitted through its role in the water cycle are also referred to as *ecosystem benefits* (cf chapter 6.1 and 6.4); however, they have received much less emphasis in the economic literature than other kinds of benefits. One notable exception is the economic valuation of groundwater-dependent wetlands, which will be introduced briefly below in 3.1.1 and discussed further in chapter 6.4. Otherwise, the lack of systematic empirical evidence limits an exhaustive discussion of ecosystem benefits.

It should also be noted that this survey will mainly be restricted to the *qualitative* aspects of groundwater protection, which is not to belittle the importance of quantitative problems. At the same time, the two are clearly interdependent: in some cases quantitative measures will be required to reach qualitative targets; therefore the quantitative aspects of groundwater will also be addressed on some occasions, e.g in connection with shallow aquifers or saline intrusion (cf. chapter 3.3.2 and 3.3.3).

3.1 Sources of pollution

The following section surveys groundwater pollution problems in different economic sectors; it is intended to set the scene for a more in-depth discussion of economic aspects in the ensuing chapters. The sector covered most extensively in the literature is agriculture as the main source of diffuse pollution. Particularly in the fields of nitrate and pesticide pollution, there is a range of articles on the costs and benefits of groundwater pollution and its remediation. Somewhat less attention has been devoted to pollution from other sectors; in this context, industry, mining and transport will be discussed.

3.1.1 Agriculture

Agriculture is among the sectors that contribute most to groundwater pollution, and in particular is one of the main source for diffuse pollution. The main pressures that the agricultural sector exerts on groundwater is through widespread fertiliser and pesticide application, leading to nitrate, cadmium, phosphate and pesticide concentrations in the groundwater. Concerning nitrogen applications, Lee and Nielsen (1987) find that its application in the US has increased 11-fold between 1950 and 1980, while pesticide applications have tripled in the twenty years from 1964 to 1984. For the case of France, Henin (1980) estimates that two-thirds of nitrate contamination in drinking water can be attributed to agriculture. In recent years, agricultural nitrate inputs into the soil have stabilised, albeit at a high level (EEA 1999). A different example of agricultural pollution is that of irrigation-induced salinity, which may also affect the underlying groundwater. All these are typical examples of diffuse pollution: although nitrate or pesticide concentrations in the topsoil are not extremely high at any particular point, it is the masses of the polluting substances that are applied over long periods which lead to pollution.[11] For cases of diffuse pollution, the potential for remediation is limited, since the polluted area cannot be isolated and contained. Therefore the prime focus must be to reduce the input of nitrates and pesticides.

In the case of agriculture, the analysis of pressures on groundwater quality is complicated by the fact that they do not only depend on the amount and kind of fertiliser or pesticide applied. The effect also depends on the crops planted, the irrigation management system, the meteorological conditions, the timing of the application, the geological and hydrogeological conditions, and on soil conditions (such as the amount of nitrates already contained in the soil). Consequently, diffuse agricultural pollution can be reduced not only through reduced applications of fertilisers and pesticides, but also through accompanying measures such as improved irrigation management, timing of the applications, choice of crops and crop rotation, and by adapting application levels to local soil conditions. Many of these improvements can be achieved at little cost through information and support of farmers, since there is still some room for improvement of current farming practices. These issues will be discussed in more detail in chapter 4.1.

At the same time, the impact of agriculture in terms of groundwater pollution is not limited to the “usual suspects”, nitrates and pesticides. Gardner and Yong (1988) present a case where flawed irrigation management has lead to a high salinity in the upper Colorado River basin. They estimate that about 37% of the dissolved salt in the Colorado river can be traced back to irrigation practices. Excess irrigation leads to salts leaching from the soil into an underlying aquifer with extremely high salinity (10.000 mg/l), from where they contaminate the river water. As the key to reduced leaching lies in improved irrigation management, this case is an interesting example of the interconnectedness of quantitative and qualitative aspects of groundwater protection.

3.1.2 Industry and Mining

Groundwater pollution from industry and mining is typically point-source pollution, which means that it is typically limited to a identifiable number of sites with high concentrations of the contaminant; nonetheless, the impact of each individual incident may be very different depending on the hydrogeological conditions and the contaminant mix. Point-source pollution from industry occurs mainly through accidental spills of dangerous substances (hydrocarbons fuels, soluble toxic substances used as inputs to the production process, such as solvents), and through improper management and disposal of industrial waste.

A typical case of point-source pollution from industry and mining are contaminated sites. These are locations where hazardous substances, such as hydrocarbons or industrial waste, have been improperly disposed. Contaminated sites can be found at military, commercial or industrial sites, including oil production and storage, mining, and other industries that use hazardous materials. In the case of mining, problems arise from mine dumps containing chlorides and heavy metals, from acid drainage and other contamination from mine tailings. Many such problems have resulted from disposal methods that were considered standard practices at the time. At the same time, groundwater pollution from mining is not restricted to mine dumps: in many cases of disused mines, the pumping wells that were used to keep the

[11] There may also be examples of point source pollution from agriculture, e.g. in the case of inappropriate storage and accidental spills of hazardous substances, underground storage tanks etc., but these are not typical for the sector.

water level low are switched off. As the groundwater table rises, minerals, heavy metals and acids are washed out from the abandoned mines. For groundwater pollution, these contaminated sites represent an continuous source of pollution unless remedial action is taken.

A particular problem in the context of contaminated sites is that of historical pollution and orphan sites. Particularly in the case of mines, it may no longer be possible to hold anyone responsible for the cleanup of contaminated sites, since the responsible firms ceased to exist. Balkau (1999) argues that in many cases of historical contamination, familiarity with the contamination has increased to the point where the existence of a problem is even denied; instead it is assumed that the historical situation has stabilised. This may be misleading, since many historical sites cause ongoing contamination of the underlying aquifer.

A separate case of industrial point-source pollution stems from the dangers from leaking underground storage tanks, which are used for storage of hydrocarbon fuels or other substances with a potentially damaging effect on groundwater.[12] Since underground storage tanks for private or commercial use are widespread over Europe, the pollution from leaking underground storage tanks is especially hard to monitor and control.

3.1.3 Transport

There are five main channels how pollution from transport affects groundwater quality:

- through spills of petroleum or hazardous materials as a consequence of accidents;
- through leakage of oil or petrol from badly serviced engines;
- through the application of road salt;
- through the application of pesticides to roads and railway tracks for weed control; and
- through air-borne emissions from traffic that enter the aquatic cycle.

The Danish Environmental Protection Agency recently conducted a survey of the costs of groundwater and soil pollution from road transport (Miljøstyrelsen 2002). Unfortunately, the project – being intended as a first preliminary survey only – revealed that some central pieces of information for a full economic valuation are lacking; in particular, the quantitative relations between emissions and exposure, and between exposure and damage, are poorly explored. The authors therefore conclude that further analyses would have to be restricted to the most prominent kinds of damage.

Ojala (2000) has researched the groundwater-related impact of petroleum leakage and spills of hazardous materials as a consequence of road accidents (cf. Box 7.3 for an extensive discussion of her findings). In her analysis, she considers different options for the protection of groundwater resources from such spills, ranging from simple administrative measures such as improved road signing in areas with a high groundwater vulnerability, to protective measures in the design and construction of new roads.

Finally, a survey paper by the Canadian Victoria Transport Policy Institute (2002) quotes a range of studies that have assessed the effects of pollution from transport on water; of these, two are related at least partly to groundwater. Delucchi (2000) considers leakage from fuel storage tanks, large accidental oil spills as well as oil leakage from motor vehicles. He finds that water pollution from these sources in the US imposes annual environmental costs of US$ 0.4 to 1.5 billion, which equals about 0.03 US¢ per kilometre driven. Unfortunately, there is no assessment how much of this pollution affects groundwater. Somewhat lower results have been reached in research done by the British Columbia Ministry of Transportation and Highways, which estimate the water pollution and hydrologic impacts of road transport at 0.02 €¢ per kilometre.

---

12 The issue of underground storage tanks is not restricted to industry alone, but also applies to private households, or indirectly to the transport sector (filling stations).

3.2 Pollutants

In order to categorise the different forms of groundwater pollution, one way is to group them by pollutants. This categorisation overlaps partly with the sectoral approach presented in the previous section: many pollution problems are directly associated with a special sector; for example, nitrate pollution is almost exclusively caused by agricultural fertiliser applications. The different pollutants can also be roughly grouped in the two main categories of groundwater pollution, namely point source pollution, or pollution from diffuse sources. Most of the pollutants described here belong to the first category.

3.2.1 Nitrate

Nitrate (NO_3^-) is an important plant nutrient, as plants take it up directly from the soil. Also, in the process of nitrogen fixation, bacteria and blue-green algae convert atmospheric N_2 to forms that plants can absorb through their roots (ammonia and nitrate). Nitrates can be returned to the soil through animal urine, feces, carcass decay, and plant decay. Natural nitrate levels in groundwater are generally very low (less than 10 mg/l NO_3).

Nitrate charges into groundwater are mainly caused by diffuse sources, specifically through the use of synthetic and organic nitrogen fertilisers. The intensification of agricultural activities has often resulted in significant over-fertilisation. Surplus fertiliser is not taken up by the crop, and when this exceeds the soil's buffering capacity, nitrate is leached from the soil into the groundwater. Industrial emitters contribute to pollution through atmospheric emissions of nitric oxide and nitrite, mainly from energy production. Municipal sources are the disposal of municipal sewage by sludge spreading on fields, or the infiltration of insufficiently treated waster water from treatment plants into an aquifer. In addition, there are some point sources of N pollution, such as poorly designed landfills or leaking sewerage systems.

Nitrate metabolites directly react with haemoglobin in human blood and other warm-blooded animals to produce methemoglobin. This impairs the ability of red blood cells to transport oxygen. This is especially toxic to babies under three months of age, for whom it may lead to a potentially fatal condition that is referred to as "blue baby" syndrome. Similarly, nitrate might also play a role in the formation of carcinogenic nitrosamines.

Total application of nitrogen fertiliser in Europe, used as an indicator for the pressure on groundwater, is stable at a high level. In the years 2000 and 2001, annual nitrogen fertiliser use in Eastern Europe was at 2.950 thousand tonnes, while in Western Europe 9.340 thousand tonnes were applied. Nitrogen fertiliser application varies significantly within a country as well as between countries. Germany, for example, uses more than 100 kg/ha nitrogen fertiliser, whereas Dutch farmers use more than 190kg /ha (EEA 1999). Due to the already high nitrogen deposits in the soil, the entire area of Denmark, Luxembourg and the Netherlands is classified as a vulnerable area under the EU Nitrates Directive.

3.2.2 Ammonium

Ammonium is a nitrogen compound. It is a product of the biological decomposition of organic nitrogen compounds (e.g. animal and vegetable proteins). In aquatic ecosystems, micro-organisms transform ammonium into nitrates. Ammonium, as other nitrogen compounds, strongly boosts eutrophication processes.

Anthropogenic sources for groundwater contamination are mainly leaking or nonexistent sewerage systems or fertilisers washed from the soil. Natural high ammonium concentrations are common in oxygen poor groundwater, e.g. under bogs. Under these conditions, ammonium can not be oxidised. Health effects on humans caused by ammonium are not known.

3.2.3 Phosphorous

Phosphorus (PO_3) is an essential element for the growth and metabolism of plants and animals. The salt of the phosphoric acid is used as fertiliser; in the past, it was also a component of cleaning supplies. The concentration in water is generally no more than 1mg/l. Phosphorus is the main cause of eutrophication and the respective deterioration of water quality. Excessive phosphorus causes rapid growth in photosynthetic aquatic life such as phytoplankton and macrophytes. Resulting explosive algae growth and decay leads to severe oxygen shortages in surface and groundwater. For the human organism, phosphorus is practically non-poisonous and, in small amounts, even an essential nutrient.

A natural source of phosphorus is soil erosion. Heavy rain can wash out natural deposits; this effect is aggravated by human activities like deforestation. Anthropogenic sources include human waste, industrial waste, the drainage of wetlands as well as agricultural activities. The main source of phosphorus in Europe, however, is not agricultural, but domestic and industrial waste water. In France, for example, the phosphorus produced by agriculture accounts for only 23% of total discharges. Some other sources of phosphorus inputs into streams are point sources such as outdated sewage treatment plants and rural households with defect septic tanks. Industrial phosphorus discharges stem from the disposal of organic waste (food waste from processing plants), cleaning detergents and phosphoric acid industrial cleaners.

To a great extent, the reduction in phosphorus discharges in recent years is related to major efforts to improve the processing of domestic waste water, reduce industrial discharges, and limit the use of phosphorus in cleaning supplies. The agricultural sector, on the other hand, still has much potential for further phosphorus emission reductions, e.g. by adopting better environmental management practices (Strosser et al. 2002).

3.2.4 Chlorides

Chlorides are chemical compounds (salts), containing negatively charged chlorine. They are very common and are naturally present in surface waters and groundwater, and are also found in waste waters. Chlorides are difficult to eliminate from waste waters; both septic systems and wastewater treatment plants are unable to remove them. In addition, they are able to mobilise heavy metals from soil. Chlorides are toxicologically harmless for humans, but they affect the taste of groundwater and thereby make it unsuitable as drinking water.

With regard to anthropogenic sources, chloride enters the hydrological cycle via chloride-containing liquid and solid waste (e.g. human and animal sewage, industrial effluents from the chemical, galvanic and paper industries, water softening plants, petroleum refineries, landfill leachate) and fertilisers containing chloride. In the northern and mountainous parts of Europe, groundwater contamination by chlorides often results from the storage and application of road salt for de-icing streets and highways.

Irrigation may also lead to an accumulation of chlorides in groundwater. If water from deep groundwater aquifers with a high chloride concentration is used for irrigation, this may increase the salt content of upper groundwater aquifers through infiltration. In addition, irrigation of dry soils may raise the local groundwater table, which leads to a dilution and upward movement of salts from the deeper soil to the root zone.

3.2.5 Pesticides

Pesticides are substances used for preventing, destroying or controlling any pest and unwanted species of plants or animals which may cause harm during the production, processing, storage, transport, or marketing of food, or agricultural commodities (FAO, 1990). Pesticides are used in agriculture, horticulture and forestry. For these agricultural uses, they are another example of a diffuse pollution source, since pesticides are usually applied over large areas.

Pesticides can reach surface waters via air and soil pathways. Pesticides not taken up by plants or absorbed by particles are leached into adjacent surface waters or deeper soil layers. Groundwater aquifers might also be affected by bank filtration from surface waters.

The number of legally permitted and applied pesticides varies from four substances in Malta to 531 substances in Spain. The usage of pesticides per hectare of agricultural land varies widely between countries in Europe from over 14 kg/ha to less than 0,25 kg/ha. The sale of pesticides has generally decreased over the last decade (EEA 1999).

Depending on their chemical composition, pesticides are more or less toxic for humans and animals. Moreover, chlorinated compounds such as DDT tend to be accumulated along the food chain. Other pesticides have been found to affect the nervous systems of humans and animals and even have carcinogenic effects.

3.2.6 Hydrocarbons

Mineral oils are mainly used as fuel for combustion engines, for heating purposes and as lubricants. Groundwater contamination by mineral oils arises mainly from public, private and industrial activities. If mineral oils reach groundwater, aromatic compounds can disperse in water and are transported over long distances. The main point sources of hydro-carbons are particularly old industrial, military and railway sites. Leaching from old car dumps, industrial and municipal dumping sites, as well as illegal dumping and the use of used oil for stabilising streets, also lead to groundwater pollution.

Contamination through volatile aromatic hydrocarbon compounds mostly arises from improper and careless handling, and accidents with solvents and raw materials containing aromatics in industry. Polyaromatic hydro-carbons (PAH) resulting from the incomplete combustion of organic material are discharged into the atmosphere. The atmospheric deposition of PAH is not significant in terms of groundwater contamination as PAHs are adsorbed onto humic substances and clay minerals. However, PAHs have been detected e.g. in shallow groundwater under Stockholm (EEA 1999).

Hydrocarbon, especially the aromatic compounds have been found to be carcinogenic in many cases. Other health effects comprise acute and chronic toxic effects, in particular neurotoxicity and other negative metabolic effects.

3.2.7 Heavy metals

Heavy metals include zinc, copper, chromium, nickel, cadmium, lead, and mercury. They occur in bounded and solved form. The main sources of heavy metal compounds are discharges from chemical and metallurgic industries, the application of fungicides in agriculture, the combustion of fossil fuels, mining activities as well as improper handling of waste containing heavy metals. The load of heavy metal inputs in the big rivers in Europe has decreased in recent years, mainly as a result of better environmental regulation as well as structural changes. Still, heavy metals pose a considerable threat to water resources as they enter natural systems via many pathways, are widely distributed (via air transport) and are highly persistent since they are not degraded by natural processes.

In little quantities, some heavy metals (zinc, copper, chromium) are essential for human health. Others are highly toxic even in smallest concentrations (mercury, cadmium, lead). In higher concentrations, all heavy metals can impact growth and metabolism processes. Since heavy metals are usually not metabolised, they accumulate in fatty tissue. Through the process of bio-magnification, humans are especially at risk as heavy metal concentrations tend to increase with each level of the food chain.

3.3 Local Groundwater Conditions and Pollution Hotspots

As noted in chapter 2.4, the effects of groundwater pollution are largely site-dependent, which means that the same amount of a pollutant can have very different effects on groundwater in different places, depending on the geological and hydrogeological conditions, ecosystems and surface water bodies depending on the aquifer, as well as potential economic uses. One way of approaching this problem is to focus primarily on areas with a higher risk of contamination due to a higher vulnerability. In the following, four exemplary cases will be introduced that have received particular attention in the literature: these are groundwater-dependent wetlands, shallow aquifers, aquifers in coastal areas as well as catchment areas, where aquifers are recharged from the surface.

3.3.1 Wetlands

Wetlands have received particular emphasis from economists and ecologists because of the plenitude of ecological services they provide. Turner et al. (2000) list among these:

- flood control;
- groundwater recharge;
- nutrient removal;
- retention of toxic substances, and
- maintenance of biodiversity.

Numerous economic sectors make use of these services, such as agriculture, fisheries, forestry, water supply and recreation. In addition, wetlands are of special interest because they function as an interface between different ecosystems: they link ground- with surface water bodies as well aquatic and terrestrial ecosystems. This also makes wetlands susceptible to groundwater pollution: although wetlands themselves form part of surface water, groundwater-dependent wetlands can be seriously affected by polluted groundwater. These aspects will be dealt with in more detail in chapter 6.4 under the heading of ecosystem benefits; unfortunately the links between groundwater pollution and wetland functions have not been researched well in most cases. The following examples highlight some of the aspects that are economically most relevant for the relation of wetlands and groundwater.

Box 3.1: Some Economic Aspects of Dealing with Wetlands and Groundwater

Turner et al. (2000) argue that wetlands have been damaged or converted all across Europe because of a lack of information on the sides of decision-makers. There may have been some examples in the past of wetland conversions that were socially and economically beneficial, especially where the returns from competing land uses were high. Yet in plenty of other cases, it appears that wetland conversions have brought little benefit to, or have even imposed a net cost on society. In Turner et al.'s view, this is because both public decision-makers and economic agents lack understanding of the multitude of values that may be associated with wetlands, as well as the complex interrelation between the various environmental services that they provide. In this way, wetlands suffer the consequences of both a market failure and an intervention failure.

Byström (1998) evaluates one particular service of wetlands, namely the retention of nitrates. Through sedimentation, uptake in biomass and denitrification, wetlands can absorb nitrates from surface- and groundwater. In contrast to Kosz (1996) and Turner et al. (2000), Byström is not only concerned with the preservation of existing wetlands, but also with the creation of new ones. In a similar fashion, Söderqvist (1998) analyses the creation of new wetlands in Southern Sweden from an economic point of view. He presents a cost function for wetland construction, which comprises construction cost, maintenance cost and the opportunity cost of the land used, and compares this with the estimated benefits of wetlands. In considering the latter, however, the benefits are only reduced to nitrate retention, while amenity values are recognised but not quantified. The approach of valuing wetlands according to their potential for nitrate retention adds an interesting twist to their valuation: it is not a damage to the wetland as a consequence of pollution which is considered, but instead the value of a wetland in cleaning up the pollution.

The examples and considerations above show that wetlands play several important roles in the context of groundwater protection and pollution: wetlands are among the ecosystems that are most likely affected by the discharge of groundwater from a polluted aquifer. At the same time, the retention of toxic substances and the removal of nutrients is one of the ecological services that wetlands provide. This raises an interesting, but complex trade-off:

constructed wetlands can be used as a way to prevent further pollution of an aquifer from surface water with a high nitrate content. In this way, they are primarily an instrument for surface water protection, and enhance groundwater protection only indirectly. For moderate concentrations, the case studies by Byström and Söderqvist show that this is a cost-efficient and competitive method of nitrate removal – possibly even more efficient than the reduction of nitrogen inputs by other means. At the same time, nutrient removal and retention of toxic substances are no panacea – excessively high concentrations will eventually impact the functioning of wetlands. These issues will be elaborated further in chapter 6.4.

3.3.2 Shallow Aquifers

Shallow aquifers are particularly vulnerable to contamination, since there is only a thin layer of sediment to protect the groundwater body – which is sometimes less than a meter below the surface. As a consequence, emissions on the surface are more easily transported to the groundwater; on the other hand, a larger share of the contaminated groundwater feeds to surface waters. While this raises the probability of adverse effects on dependent ecosystems, it means that contamination episodes in shallow aquifers are likely to be more short-lived than in deep aquifers where there is less exchange with surface water bodies.

Hellegers et al. (2001) present an analysis of groundwater pollution in the case of shallow aquifers. The model they design, using the Netherlands as an example, aims at combining qualitative and quantitative aspects of groundwater pollution. This is mainly done in two ways. First, by incorporating a "dilution effect", which uses the fact that the inflow of a pollutant will lead to a more severe contamination if it is absorbed in a heavily-depleted groundwater body. Secondly, they use the pollutant concentration in the recharged ground water and in the water abstracted from ground water as two separate variables, where groundwater quality can be improved if the abstracted water is more contaminated than recharged water. One main finding Hellegers et al. arrive at is that the internalisation of externalities from groundwater pollution is particularly important if the recharge of groundwater is large relative to the stock.

Although the analysis by Hellegers et al. is purely model-based and in that sense theoretical, it does add more clarity to the interrelation of qualitative and quantitative aspects of groundwater pollution. Obviously, the effect of a given pollutant entry will be the more severe the smaller the groundwater body is, and it will last the longer the less exchange of water takes place with surface water bodies.

3.3.3 Coastal areas

In coastal areas, aquifers are frequently threatened by sea water intrusions into the aquifer; this problem is referred to as saline intrusion. It is an example of the interconnectedness of qualitative and quantitative aspects of groundwater protection: only if aquifers in coastal areas are over-exploited is there a chance for saltwater to intrude into the unsaturated zone. In some parts of Europe, saline intrusion is one of the main causes for groundwater quality degradation in coastal areas. One peculiarity of saline intrusion is that the quality degradation is not due to man-made pollution. Therefore, to improve or maintain water quality the *abstractions* must be limited - rather than restricting emissions of some sort, as in other cases of groundwater pollution. This means that one solution is improved water management, in order to optimise uses and reduce abstractions of groundwater.

Box 3.2: Three Approaches to Measuring Economic Aspects of Saline Intrusion

Moreaux and Reynaud (2002) construct a dynamic, spatial model of saline intrusion into a coastal aquifer. Because of its spatial component, the model describes the optimal water management policy both in terms of the location of wells, and the volume of abstracted groundwater. Moreaux and Reynaud use water prices as the central instrument for influencing the volume of groundwater abstraction; one of their central points is that these water prices have to be differentiated to reflect the local circumstances. They assume that the cost of saline intrusion equals the increased cost for providing clean drinking water. Since saline intrusion first occurs at the margin, the drinking water wells closest to the sea have to be closed first, and water pumped from wells further inland. The higher cost of drinking water provision consists of the drilling costs for new inland wells, and of the cost of pumping the water to the coast. These costs can be seen as an externality imposed on coastal dwellers by the inland communities. The optimal tax level then depends not only on the available resources and the demand function, but also on the population distribution between the two regions.

The economic implications of saline intrusion were researched by Stéphane Robichon (Agence de l'eau Adour-Garonne 2002). The study investigates different options to prevent further groundwater quality degradation in the Gironde Département. The aquifers in this area are affected significantly by constantly high levels of water abstraction. Over three quarters of these result from human consumption, since the aquifers supply the city of Bordeaux and its surroundings, as well as the coastal areas with a large tourist industry. The remaining abstractions are used for irrigation (17%) and industrial uses (8%). The massive abstractions have lead to a pressure drop in the water table, which in turn leads to the inflow of brackish water from the Gironde estuary, as well as the inflow of polluting substances such as nitrate and pesticides from the surface. Robichon considers the use of different measures in order to achieve a given abstraction reduction, including demand management through differentiated water prices and informational measures, as well as changing to water supply from alternative sources.

Aguilera Klink et al. (1998) discuss problems arising from groundwater overexploitation for the Spanish island Tenerife; unfortunately they do not report costs and benefits of different groundwater services or groundwater protection measures. They list as major problems the intrusion of saltwater into the aquifer under Tenerife, caused by overexploitation of the aquifer; the gradual deterioration of the aquifer through pesticides and nitrates from agriculture; and the energy use associated with desalination of brackish water and salt water. They argue that the cheapest response to these problems is through improved water demand management, which leaves much room for improvement at the moment. Options include increasing the technical efficiency of water distribution networks and applications, and the reform of water rights, which were previously established on a "first-come-first-serve" basis. However, the overriding option in their view is to open a public debate on water use, in order to arrive at an adequate collective understanding of the problems related to groundwater protection. The authors argue that thereby it may be possible to arrive at a water demand management based on two insights about the economic use of water: that "it is cheaper to save a cubic meter than to produce another one"; and that "harnessing the work of natural processes [...] by replacing human effort with the economics of nature is the essence of good sense, both economically and ecologically" (ibd., p. 25).

Therefore, as in the case of shallow aquifers, coastal aquifers threatened by saline intrusion again show that the quality of groundwater in many cases cannot be treated separately from quantitative aspects. One central conclusion of the literature dealing with episodes of saline intrusion is that the improved water management is most likely part of any efficient solution, thereby combining the stabilisation of groundwater levels as a quantitative target with the prevention of further degradation as a qualitative target.

3.3.4 Recharge and Catchment Areas

Another kind of areas where pollution is likely to have significant adverse effects on groundwater are the recharge and catchment areas; consequently, these areas have received much emphasis in the literature.

- Recharge areas: how much surface water percolates through the topsoil and enters an aquifer, and how long this takes, depends on the local geological conditions. In areas with a highly permeable the topsoil layer, failure to implement strict groundwater protection measures is more likely to lead to contamination of an underlying aquifer.

- Catchment areas: the catchment area is that part of a groundwater recharge area that is the source of water for a well.[13] The areas surrounding larger wells and waterworks typically have a special protected status, in which certain kinds of land uses are restricted. In these catchment areas, groundwater protection is especially relevant because contamination would directly affect groundwater intended for use as drinking water.

The two approaches differ in so far as the need for special protection is determined by the geological and hydrogeological conditions in the former case, whereas in the latter case the central factor is the current use of the aquifer, or the intention for future use. In practice, both criteria are likely to overlap. One way of combining the two is through *risk-based* groundwater management (cf. chapter 7.2.1): in this approach, recharge areas are regarded as areas with a high probability of adverse effects, whereas catchment areas are seen as cases where there is a large potential impact from pollution. Economic studies in these fields are mainly concerned with the question of how use restrictions in the protected areas affect the income of the affected parties, e.g. if farmers have to change their agricultural practices in the protected areas. Some experiences will be discussed in the context of cooperative agreements, which are discussed in chapter 4.1.3.

[13] Catchment areas are also referred to as the *contribution zone* or *capture zone* of a well.

4 Instruments for Groundwater Protection and Remediation

The following two chapters will both address the protection and remediation of groundwater, albeit from a different angle. This chapter discusses different economic and non-economic instruments for the protection of groundwater from pollution (4.1) and for the remediation of contaminated groundwater (4.2), and will then offer some general considerations on the choice of the instrument mix under different circumstances. Chapter 5 builds on this and extends the analysis by presenting some empirical evidence on the cost of protection and remediation measures. However, because the empirical literature on this issue lags behind the theoretical knowledge, not all of the instruments discussed in chapter 4 will also be dealt with in chapter 5. Additionally, following the structure used in the literature, chapter 5 will be structured around different types of pollution, whereas this chapter is structured around the different instruments and their strengths and weaknesses.

This chapter comprises two different approaches – *protection* as a proactive instrument to prevent pollution from occuring will be discussed in chapter 4.1, whereas chapter 4.2 addresses *remediation* as a reactive approach that aims to control, and, if possible, eliminate groundwater pollution that has already occured. In practice, both activities will often be pursued at the same time, and some measures may qualify as protection and remediation measures.

4.1 Instruments for Groundwater Protection

There are a number of ways to achieve a given environmental target. The instruments that can be used to influence the behaviour of consumers and producers towards less environmentally harmful behaviour range from informational measures to direct regulation, which bans certain behaviour. Economic instruments, which aim at influencing behaviour by changing the economic incentives that producers and consumers face, are gaining in relevance. A range of instruments have been applied successfully to groundwater protection in the past, in other cases the reform of existing instruments will be helpful or necessary to improve groundwater protection. This section will give an economic overview of the experience with some of the different instruments and measures that have been used in the past, and will discuss potential further developments. Since the available literature is mainly concerned with pollution from agricultural sources, the emphasis will be primarily on diffuse agricultural pollution.

4.1.1 Taxes and Subsidies

Taxes are one of the economic instruments most commonly used to limit environmentally harmful behaviour by individuals or firms. The theory of environmental taxation with the aim of increasing overall economic welfare can be traced back to 1920, when they were first proposed by A.C. Pigou. However, in the field of groundwater protection, environmental taxes are not used widely; some examples are listed below. Interestingly, taxes on groundwater-related pollution are much more common in the Candidate Countries than they are in the existing Member States. A topic that is closely related to environmental taxes is that of subsidies; these will be discussed further below.

Of the existing examples of taxes with an impact on groundwater pollution, most are related to agricultural diffuse pollution (Source: OECD 2002):

- Taxes and charges on *nitrogen fertilisers* are used in the Netherlands, Sweden and Denmark; Sweden also levies a tax on cadmium contained in fertilisers.
- *Pesticides* are taxed in Belgium, Denmark, Finland and Sweden.
- Charges on *wastewater* are common throughout the EU, but are only indirectly relevant for groundwater protection.

- Some countries have introduces taxes or charges for *groundwater abstraction*; this is the case in the Netherlands and France, and some German Länder. However, abstraction taxes or charges are more relevant for the *quantitative* management of groundwater protection, and therefore again only indirectly relevant for the protection of groundwater quality (cf. Box 4.2).

Box 4.1: Efficiency of Taxes on Diffuse Source Agricultural Pollution

In order to assess the economic impacts of nitrogen taxation, Johnson et al. (1991) simulate the effect of two different forms of nitrogen taxes on farmers in the Columbia basin in Oregon. They find that the price elasticity for nitrogen demand is very low, so that high taxes on nitrogen inputs do not lead to sizeable reductions in nitrogen applications. In a second simulation they conclude that taxes on nitrogen leachate (rather than on nitrogen input) are more efficient at a lower cost , which they estimate as 3-6% of total profits. However, such Pigovian taxes on nitrogen pollution are hard to implement in practise because of the difficulties in monitoring nitrogen leaching.

Gardner and Young (1988) present an application of taxes to a different kind of agricultural pollution. In the case they investigate, excess irrigation leaches salts from the topsoil into an underlying aquifer, leading to an extremely high groundwater salinity of 10,000 mg/l, and consequent pollution of the Colorado river. In their comparison of different instruments, they assume that six irrigation technologies are available that differ in their total costs, labour intensity and efficiency. They compare the following instruments in terms of their effect on irrigation technologies:

(a) the existing system of subsidies for new irrigation technology,

(b) a (hypothetical) tax on actual salt leaching induced by farmers, and

(c) a tax on water used for irrigation.

Their conclusion is that the emission tax (b) is clearly the most efficient instrument. Taxing irrigation water (c) also achieves a reduction of salt discharge, but leads to high costs for the farmers. Subsidies for irrigation improvements (a) lead to decreasing salt discharges and increasing farm incomes at the same time, but at the price of high social costs.

The findings from the analysis by Gardner and Young can be generalised to other cases of diffuse source pollution: essentially, they find an emission tax to be most efficient because it is targeted directly at reducing emissions (i.e. salt discharges) in opposition to the other two instruments, which function indirectly. However, the measurement and monitoring problems associated with the emission tax make it an impractical instrument, as the authors admit; it should thus rather be seen as a hypothetical benchmark. Instead, a tax levied on crucial inputs with potential adverse effects may be a less efficient, but more practical solution. Although this finding is derived in the particular context of chloride pollution, it can be extended to other types of diffuse pollution as well.

Box 4.2: Groundwater Consumption Taxes and their Use

Neumüller (2000) presents an extensive evaluation of groundwater taxes in the German Bundesland Hessia. The analysis describes consumption taxes only, which are more relevant to quantitative aspects of groundwater protection; however, qualitative aspects can be addressed through the use of tax revenues. In the analyses of groundwater taxes, Neumüller distinguishes between a ‘push‘ and a ‘pull‘-effect.

- The push-effect arises from higher prices, which pushes consumers to reduce their use, or switch to other sources.
- The pull-effect, on the other hand, arises from measures undertaken with the revenue from taxation: typically, these are government programmes that aim at inducing different behaviour, as well as investments in groundwater protection.

The author argues that, because demand for groundwater is fairly inelastic (i.e. it does not react strongly to price changes), the push-effect of groundwater taxes is likely to be limited – meaning that it will bring revenue, but not induce many changes of behaviour.[14] Also, the push-effect is likely to be outweighed by the pull-effects, especially in the medium to long run. Tax-financed government measures apply both to quantitative and qualitative aspects; among the instruments to improve groundwater quality are land acquisition in groundwater protection zones, information and consultation of farmers, construction of manure storage facilities and restoration of wastewater canals. Unfortunately, Neumüller does not assess the effectiveness of these instruments, not least because they have only played a subordinate role in comparison to quantity-oriented measures.

In the context of taxation, a related issue that needs to be considered is that of subsidies. From an economic perspective, subsidies are effectively negative taxes: the steering effect, which makes taxes such a versatile instrument by changing the incentive structure, can also be realised through an efficient targeting of subsidies. The relevance of subsidies in groundwater protection is most evident for the agricultural sector: agriculture is both a main contributor to groundwater pollution in many parts of Europe, and a major recipient of subsidy payments.

Box 4.3: Increasing the Efficiency of Agricultural Subsidies

Kim et al. (2000) present an analysis how the efficiency of existing subsidies for agricultural irrigation systems can be improved in an economically efficient way. They argue that in many cases, agricultural irrigation is managed fairly inefficiently even in industrialised agriculture, and that existing US government subsidy programmes promote wrong and inefficient types of irrigation.

For the study area in Central Nebraska, they compare the use of four different irrigation technologies (conventional furrow irrigation, tail-water recovery irrigation, surge-flow irrigation, and centre-pivot irrigation). These irrigation systems are compared in terms of their private costs and their effect on nitrate levels in groundwater. Concerning the private costs, the authors find that in the absence of any government support, and blending out effects on nitrate contamination, the adoption of a surge-flow irrigation system would be the most efficient choice. This is the case because the increased irrigation efficiency compared to other technologies would compensate for the additional costs.

However, taking the effect of irrigation on nitrate concentrations into account, they point out that centre-pivot irrigation is the only technology that achieves a reduction of nitrate contamination levels (of about 11%), whereas all other technologies would lead to a concentration increase of up to 40% over 15 years. Kim et al. therefore conclude that the current cost-sharing system, whereby farmers receive targeted support for the adaptation of new irrigation technologies, should be extended to centre-pivot systems. They calculate that a government cost share of 22.55 US$/ha would be justified in support of the adaptation of centre-pivot-systems. Compared to the existing system, these subsidies would be more efficient economically because of their positive effect on nitrate contamination.

If agricultural activity leads to a deterioration of groundwater, this suggests that polluting activities themselves may also be subsidised. This would be a case of a “perverse subsidy”, where a public payment actually works against the public interest rather than promoting it. In this case, reducing or redirecting subsidies would have the same effect on the incentive structure as a tax, but at a lower administrative cost. Therefore, rather than taxing nitrate fertiliser or pesticide use, a first step would be to assess whether existing agricultural subsidies take sufficient account of groundwater pollution, and if not, correct them accordingly.

---

[14] For example, Neumüller finds that the current groundwater taxes of 0,05 c/m^3 implemented in some German Bundesländer are not likely to have any effect at all on consumption.

In connection to the ongoing discussions about the reform of the EU Common Agricultural Policy, these approaches are discussed under the heading of *cross-compliance*: the basic idea is to make subsidies partly conditional on compliance with good agricultural practice standards. This instrument is not entirely new, but so far was only an optional measure and has therefore only been implemented in a few Member States, and with regard to selected practices.[15] In this context, there is some potential for a further integration of groundwater protection requirements into the Common Agricultural Policy. For example, good agricultural practice should not only be defined in terms of nitrate or pesticide applications per hectare; in addition, it should also include other parameters that determine the effect of agricultural practices on groundwater. These might include improved timing and dosing of fertiliser or pesticide applications, taking account of hydrogeological and soil conditions; storage of fertilisers and pesticides; improved irrigation management; and crop choice and ploughing practices that minimise nitrate leaching (see Dwyer et al. (2000) for a detailed discussion).

A particular problem in relation to agricultural subsidies arises in connection with cooperative agreements and compensation schemes (cf. chapter 4.1.3). In these cases, municipal water suppliers compensate farmers for income losses associated with less intensive agricultural practices. From an economic perspective, one drawback of such compensation schemes is that agricultural output is subsidised twice. The “normal” level of agricultural output is already largely determined by agricultural subsidies. Therefore, a compensation for foregone output will partly be a compensation for foregone subsidies. This makes the calculation of economically efficient compensation levels very difficult.

4.1.2 Tradable permits

Tradable permit systems have become very popular among environmental economists in recent years, because they promise to achieve a given environmental goal with the efficiency of a free market solution. The basic idea is simple: first, the optimal amount of nature use is defined - either in terms of pollutant emissions, or in terms of resource abstraction. The definition of the desired amount of nature use itself can be based on a cost-benefit analysis, but can also be the outcome of a political bargaining process. A central authority can then sell off legal titles that permit the use of nature, typically to the larger users such as industrial or agricultural companies. It is crucial for the working of the system that a user can not emit (or abstract) more than the limit corresponding to the number of permits he / she holds; to this end, a tradable permit scheme has to be combined with an effective monitoring system and a system of penalties so as to enforce compliance. In the optimal case, such a tradable permit scheme will then guarantee that nature use is optimised: if resources were used more efficient in one company than in another, it would be profitable for the less efficient company to reduce its production or to improve its production technology, and to sell off the corresponding amount of permits. By this mechanism, the cost of abating emissions would eventually be equalised across all companies. There are a number of cases where tradable permit systems have been employed to control the pollution of surface waters, mostly in the USA, but also in Australia and Mexico[16]. In most of these cases, tradable permits were used to control point-source pollution; in a few other cases, they have also been applied to diffuse pollution sources.

To our knowledge, however, there is so far no example where tradable permit schemes have been applied to groundwater pollution[17]. This is not surprising, given the difficulties associated with groundwater pollution:

---

15 These are the UK, France, the Netherlands, Denmark, Finland and the Republic of Ireland. However, the specifications of cross-compliance schemes differ between countries, and not all of them are related to practices with an impact on groundwater; e.g., the British and Irish models are mainly used to prevent overgrazing. For an overview of existing measures, see Choudhury et al. (2002).

16 For a survey, see Kraemer and Banholzer (1999) and Kraemer et al. (2002).

17 For a theoretical treatment of the issue, see Morgan, Coggins and Eidman, 2000.

- as stated above (cf chapter 2.4), it is often difficult to relate negative impacts on the environment to particular pollutant sources, especially in the case of diffuse pollution. For a tradable permit scheme to function efficiently, however, a reliable mechanism for monitoring is indispensable. Because of the inherent difficulties, such a control mechanism would be very costly in the case of groundwater pollution, which would reduce the efficiency of the instrument significantly.
- a tradable permit scheme equalises the cost of emissions throughout the trading area. This may be optimal from an economic efficiency perspective, but not so from a social welfare point of view. This is because emissions in different circumstances (temporal, spatial, seasonal, hydrological, meteorological) will have very different effects on the groundwater, and will therefore impose different external effects on groundwater users. These differences can be captured in a taxation scheme or in command-and-control regulation, however to include them in a tradable permit scheme would deprive the system of its transparency and its efficiency.

Box 4.4: Qualitative Aspects of a Possible Tradable Abstraction Rights Scheme

Kahlenborn and Klaphake (2001) evaluate a suggested scheme for tradable abstraction rights in Germany, referring also to the expected impact such a scheme would have on groundwater quality. The scheme, which was discussed by the German *Sachverständigenrat für Umweltfragen* (expert advisory board on environmental matters), consists of regionally bounded trading systems, where the withdrawal rights for one particular groundwater body are auctioned off to the highest bidder. At the same time, the existing system of payments for groundwater extraction are abolished. In evaluating this proposal, Kahlenborn and Klaphake are concerned that the drive towards cost-efficiency inherent to a tradable permit system would result in a very selective groundwater protection. They argue that a permit scheme would primarily lead to the exploitation of those groundwater bodies which are easily accessible and little polluted, hence require little cost for abstraction and treatment. However, aquifers which are already polluted will not profit from such a trading scheme. In addition, because any tradable permit scheme presupposes scarcity of the traded good, Kahlenborn and Klaphake even argue that local authorities might abstain from restoration and tolerate declining groundwater quality levels for some aquifers, since a decline in quality would raise the value of the remaining unpolluted resources.

As becomes clear from the arguments and examples above, tradable permits are not the instrument of choice for limiting groundwater pollution. They are, however, a possible instrument when it comes to managing quantitative targets, e.g. by limiting the total abstraction from any aquifer and selling off abstraction rights. Yet, although such tradable permit schemes are introduced for quantitative management, they will also have effects on groundwater quality: how much anyone is willing to pay for an entitlement to abstract groundwater will also depend on the purity of the groundwater.

4.1.3 Cooperative Agreements

Cooperative Agreements are a practical application of the theoretical idea embodied in the Coase Theorem (cf. chapter 2.2). For pollution problems, a common situation is that the benefit of a pollution reduction exceeds its cost. In these cases it is (theoretically) possible to negotiate a solution in which the polluter restricts the polluting activity, and is compensated for the foregone income by the beneficiary of the pollution reduction. Such agreements can but need not be concluded under the stewardship of a government. Likewise, they can either be negotiated on the local level in an ad-hoc fashion, or they can be institutionalised and regulated with legal means.

In the case of groundwater protection, cooperative agreements are typically concluded between water supply companies and agricultural polluters. There is some scope for welfare improvements from such deals, since the cost of removing nitrates or pesticides from groundwater used for drinking water purposes in many cases exceeds the cost of reducing applications of these substances. A conceptual problem with such an approach is that it conflicts with the polluter-pays-principle, a fundamental principle of European environmental policy in general and the Water Framework Directive in particular: if farmers receive compensation for reducing nitrate application, this effectively amounts to a “victim-pays-principle”.

Box 4.5: The German *Wasserpfennig*

Cooperative agreements have a long tradition in Germany. Pfaffenberger and Scheele (1990) describe the model of the German *Wasserpfennig*, which has been applied in the German *Länder* Hessia, Lower Saxony, North Rhine-Westphalia and Baden-Württemberg. The underlying idea is that farmers are compensated by water supply companies if their farming practices are impaired by groundwater protection requirements. This compensation is levied from all customers of the water supply companies; the level of compensation is determined through negotiations between farmers and water suppliers either on the municipal or on the sub-state level. The pitfall of this instrument is that, in order to calculate the level of compensation, the common standard for agricultural practices has to be defined first. For the farmers, this creates an incentive to lobby for a definition that is based on very intensive agriculture, since this will increase their compensation. However, since this also raises costs for the water suppliers – and ultimately the consumers – much of efficiency of the theoretical solution is lost in practice. A second drawback relates to the cost of monitoring: the payment of the compensation means that farmers are legally obliged to change their practices, yet the actual compliance can only be verified through extensive and expensive monitoring.

Box 4.6: The Efficiency of Cooperative Agreements in the Short and Long Run

Gramel and Urban (2001) present some theoretical and empirical evidence on cooperative agreements used to reduce nitrate concentrations in the German Bundesland Hesse. Based on interviews with farmers, they found that one obstacle was resistance by farmers who would not accept interference with their farming methods. Consequently, about a third of the payments made in the agreements went into informational measures, giving farmers agricultural expert advice about optimal fertiliser use. Actual compensation payments for farmers only accounted for slightly more than a third of total costs.

The authors remark that so far the effectiveness of the cooperative agreements has been limited: nitrate concentrations *in the soil* have decreased only in two cases, and the performance of regions where agreements had been implemented was only slightly better than in the other regions. At the same time, the authors point out that the gains from cooperative solutions will accrue in the medium to long-term only, due to the travel times of nitrates. Consequently, in order to improve the quality of drinking water abstracted from the groundwater, efforts have to begin a long time before effects can be seen. In comparison with end-of-the-pipe technologies applied to purify water intended for human consumption, this is a disadvantage, since water purification delivers immediate results and would have to be continued for some time before cooperative agreements show results. At the same time, it is equally clear that drinking water purification is only a partial solution, since it does not help to improve the condition of groundwater. Blending out the time lag problem, and potential other uses for groundwater, the authors find that cooperative agreements are an economically competitive alternative to end-of-the-pipe treatment when it comes to reducing nitrate concentrations in drinking water (cf. chapter 5.1.1 for an overview of the costs).

Box 4.7: Overview of Cooperative Agreements in the Water Sector

Heinz et al. (2002) have studied 525 cooperative agreements in the water sector. Of the total, more than 80% were concluded in Germany, followed by France and the Netherlands. The large majority (81%) of the agreements was designed to tackle nitrate pollution problems, with a minority addressed at reducing pesticide pollution or resolving quantitative problems.

In the majority of cases studied, cooperative agreements were used to prevent pollution levels from reaching thresholds where more extensive and costly protection measures would be required. They are thus primarily used as a preventive solution rather than a remediative instrument. In order to comply with drinking water standards, cooperative agreements can also be regarded as an alternative to treatment of abstracted groundwater. The agreements that were investigated in the study were either concluded directly between farmers and water suppliers, or they involved public authorities to some degree. The latter is especially likely in cases where the cooperative agreement is financed through taxes or water abstraction charges.

The authors point out that cooperative agreements are typically first accepted and by the most innovative farmers. To reach more conservative farmers as well, good communication and the inclusion of other benefits (e.g. providing expert advise to farmers) is helpful.

One central finding is that the efficiency of agreements depends mainly on their specific design: for example, the efficiency can be enhanced if compensation payments are reduced over time, as farmers successively adopt less harmful cultivation and production methods. At the same time, agreements need not be based on compensation payments only. The study also reports a number of "double dividend" cases where improved fertiliser management helped to achieve better groundwater quality, implying cost savings for water suppliers as well as higher crop yields from improved fertilisation for the farmers. In this way, cooperative agreements tend to be most efficient where they are combined with informational and other measures.

Overall, the authors' findings on the cost effectiveness of cooperative agreements are encouraging. Although most of the case studies reviewed by Heinz et al. were concluded in order to safeguard groundwater quality rather than reducing treatment costs, the costs of agreements compared with hypothetical treatment costs generally appears very favourable.

Comparing different kinds of agreements, the authors argue that preventative agreements appear to be more cost-effective and cheaper overall than remedial approaches. They also maintain that cooperative solutions *in tendency* appear to be more cost-effective than rigid command-and-control measures, especially because they can be adapted to site-specific conditions more easily.

One central aspect of cooperative agreements, the question of who has to compensate whom to what extent, depends on the legal specifications for groundwater protection and property rights (cf. the discussion of Coasean solutions in chapter 2.2). Hofreither and Sinabell (1996) discuss the differences between Germany and Austria in this regard. The authors argue that German farmers are in a quite unique position, since they de facto have the right to emit sizeable amounts of fertilisers and pesticides, and therefore have to be compensated for any agreement that curtails this right to emit. Austrian farmers, by contrast, are in a much weaker position since they do not have any similar legal basis to demand compensation. Nonetheless, cooperative agreements are theoretically possible – in this case the farmers might want to reach an agreement with water supply companies, where farmers would compensate water suppliers for accepting higher pollution levels.

4.1.4 Standards and Regulations

In contrast to the other instruments discussed above, standards and regulations do not belong to the group of economic instruments, in the sense that they do not exploit or stimulate economic dynamics or market forces. In most EU countries, standards and regulations have long been the backbone of groundwater protection instruments. In the case of groundwater protection, standards and regulations apply a.o. to:

- the definition of *good practice* standards, or the recommendation of *best available technologies* for use by firms or by private consumers,
- regulations regarding the *production, use, storage, transport and disposal* of dangerous substances,

- special regulations and protection requirements regarding industries and installations with a high potential for damaging effects on groundwater, such as landfills or underground storage tanks,
- *bans* on the use of certain substances,
- regulations concerning land use in *protected areas*.

Generally, command-and-control measures have received little attention from economists: They are usually regarded as the least efficient of all possible instruments, and therefore are typically considered as a benchmark value only. This is mainly the case because standards offer little or no incentives for behavioural changes after compliance with the standard has been achieved. In addition, they may provoke actions that run counter to the intention of the regulation (in economic terms, this is referred to as a *moral hazard* problem). For these reasons, it may be questionable whether command and control measures are the most suitable instruments to influence the behaviour of economic actors.

There are, however, some cases where command-and-control measures are clearly preferable from an economic point of view:

- Standards are regarded as the most effective instruments if pollution would lead to very large or potentially irreversible damages, and if there is large uncertainty about the effectiveness of other instruments, such as taxes (cf. the discussion of the Weitzmann theorem in chapter 2.2). This property applies mainly to point-source pollution, and here especially to contaminated sites; therefore economic instruments are not common in this context.
- Standards and regulations may also be preferable if the alternatives, i.e. economic instruments, are associated with high monitoring and transaction costs. This is especially relevant for instruments such as tradable permits, which are associated with extensive negotiations. In such a case, setting a standard would be both cheaper and faster to implement.

Box 4.8: The Costs of Groundwater Protection Areas

One of the most widely used regulative instruments for groundwater protection is the designation of protected areas, in which potentially polluting activities are restricted or prohibited. Protection of catchment areas typically takes the form of legal regulations for certain activities within the catchment area of a borehole, or the recharge zone of an aquifer. For this purpose, some EU Member States such as Germany, Austria and the UK have explicit zoning legislation which establishes protection requirements for areas at different distances from a borehole; these zones are typically based on the assumed travel times that it takes for a pollutant to reach the groundwater underneath the borehole.

The economical relevance of such approaches lies mainly in their *opportunity cost*, i.e. the foregone income of those that are restricted in their use of the area. This would mainly apply to farmers who have to cut back fertiliser applications, or industrial developers who move to different locations. In reaction to these problems, some countries have devised compensation schemes whereby the affected parties – usually farmers – are compensated for their income losses (cf. also the discussion of cooperative agreements in chapter 4.1.3 above).

However, next to these direct effects, there are also side-effects and second-round effects of catchment area protection. Second-round economic effects arise, for example, if catchment area protection impairs the production of a sector. The reduced output connotes smaller wage payments to local workers, and therefore less money spent in the region in total. However, for such effects to be measurable and significant, the affected industry must be important in the region, and it must be seriously affected (see the discussion of Messner et al. (2001) in chapter Box 7.1). A different side effect of catchment protection is that the protected areas may be associated with increased amenity and utility: if the area is afforested, it may have a higher recreational value than a maize field. Such effects make themselves felt in a higher quality of life for local residents, but they can also lead to economic gains, e.g. through increased land and house prices in the area adjacent to the protected zone.

Since the designation of protected areas has often failed to deliver the desired outcome in the past, a different strategy by water suppliers has been to buy up land in the protected areas and to use it for forestry (cf. the discussion of Pfaffenberger and Scheele (1990) in chapter 5.1.3). This approach also makes use of the fact that forests are very efficient in filtering out nitrates and toxic substances from stormwater runoff and precipitation (see the discussion of Küchli and Meylan (2002) in Box 6.5 for an economic discussion)

4.1.5 Informational Measures

Informational measures are the softest of the instruments discussed here: they cannot be used to enforce any specific behaviour, and therefore do not impose any direct costs on the regulated parties. For this reason, their effectiveness is naturally limited if they appeal to the polluters' sense of environmental responsibility only. Informational measures can be used more effectively in cases where *win-win* situations exist. These are cases in which a change in behaviour is both economically beneficial and at the same time reduces environmental pressures.

- In the case of agriculture, one possible win-win situation is to improve irrigation management: it saves money for the farmer, and at the same time it can be used to control leaching of nitrates or chloride. In other cases improved irrigation management can reduce the risk of a saline intrusion by lessening agricultural water abstractions (Aguilera Klink et al. 2001, cf. discussion of the case study of Tenerife in chapter 3.3.3).
- A different example of a win-win situation lies in more targeted fertiliser and pesticide applications (cf. Box 4.8 below). The effectiveness and the environmental effects of such applications depend strongly on the timing of the applications and on the local soil conditions: the same amount of a substance will have a very different impact, depending on the meteorological conditions, the type of soil and its saturation level. Therefore, a more targeted application of fertilisers and pesticides offers itself as a cost-effective way of reducing agricultural groundwater pollution.

Informational measures can be used as a cheap and uncontroversial way of enhancing awareness and knowledge of different technologies; it can thereby play a key role in helping to improve agricultural management practices. However, informational measures are also relevant for non-agricultural sources of groundwater pollution; by promoting a more responsible conduct by car drivers, they can help to prevent accidental oil spills etc.

Box 4.9: Reducing Nitrate Pollution at "Zero Cost"

Rejesus and Hornbacker (1999) report cases from the US where significant reductions in nitrate application and groundwater contamination could be achieved at no cost, simply by improving dosage and timing of applications. They found that in some cases farmers would actually apply fertilisers beyond the profit-maximising level, so that a reduced dose helped them to increase their profitability. Furthermore, the timing of the fertiliser application emerged to be crucial for the efficiency of applications: based on computer-based simulations for a watershed in central Illinois, Rejesus and Hornbacker calculated that the mean N pollution from corn farming was highest when fertilisers were applied in fall, and lower for spring application or a mix of both. In addition, they found that N pollution increased overproportionately if application rates exceeded the optimal recommended fertilisation rate. The authors also compared conventional application techniques, which differed in timing and total amount, with *site-specific management*, whereby the amount of N applied would take into account the productivity and other spatial characteristics of the soil. Surprisingly, this technology was only slightly more effective than some of the conventional application methods in terms of increased yield or reduced fertiliser application. They did, however, achieve their target of reducing the standard deviation of net returns, which makes it attractive for risk-averting producers.

The authors conclude that a central function of environmental policy is to provide farmers with information. This is because the costs of site-specific management are mainly learning costs for mastering the technology that have to be borne by the farmer, whereas the identification of soil conditions would be a minor expense. This shows that informational measures do not come at zero cost, since the provision and acquisition of knowledge themselves are cost factors. However, the point to note is that even the purely private benefits for farmers (in terms of reduced expenditure on fertiliser) are already sufficient to offset the cost of acquiring knowledge, while the administrational costs are relatively small.

4.2 Instruments for the Remediation of Groundwater Pollution

Groundwater remediation relates to all instruments which are used to deal with groundwater that has already been contaminated; this includes measures that actively reduce pollution, as well as controlling pollution by limiting its spread within an aquifer. The issue of groundwater remediation must be approached with a double caveat: in many cases it may not be possible to clean up contaminated groundwater, and even where it is possible, it is most likely much more expensive than preventing pollution in the first place.

Secondly, from an economic perspective, the cost of groundwater remediation can be interpreted in different ways. It can either be seen as a *cost* factor of achieving and maintaining unpolluted groundwater, or as a measure of the *benefit* of groundwater protection. In the following, the cost of groundwater remediation will be seen as a cost factor, which is balanced against the benefits of unpolluted groundwater. It thereby addresses the question whether the costs for cleaning up a polluted aquifer are justified by the benefit that a clean-up entails for groundwater users.[18]

Three different categories of response activities can be identified: restoration of an aquifer, containment of the contaminated plume or parts of it, and pollutant removal at discharge points.[19] These different options will be discussed in more detail in the following.

4.2.1 Restoration

Restoration describes the use of measures that are intended to bring a groundwater body back to its former, unpolluted state. Following a categorisation by the Flemish Public Waste Management Authority, it can be considered as a source-oriented approach, with the aim of removing the source of contamination from the polluted aquifer or soil (OVAM 2002). Whether or not it is technically possible to restore an aquifer is site-specific: for shallow aquifers with high recharge and discharge rates and therefore a high natural attenuation, restoration will be achieved more easily than for aquifers in mountainous regions, which may be shielded by many meters of solid rock, and where little or no exchange takes place with surface waters.

---

[18] The alternative interpretation, which views the cost of remediation as a measure of the benefit, is not considered. The underlying question it addresses – whether it is cheaper to protect groundwater and save on remediation, or whether to pollute first and clean up afterwards – is misleading for a number of reasons. Above all, the restoration of all polluted aquifers will hardly ever be feasible, let alone preferable from an environmental point of view. Such an approach would also be contradictory to the no-deterioration target of the Water Framework Directive. In addition, it assumes a general liability by polluters to bring a groundwater body back to its original state under all circumstances. Whether such a strict interpretation of the polluter-pays-principle could be implemented is highly doubtful.

[19] Raucher (1983) adds to these two different options: avoidance, which implies switching to another source of water, and a fourth option, which is to take no action at all.

In general, two broad types of restoration technologies can be distinguished:

- *In situ measures*: such measures treat the contaminant and the contaminated groundwater within the aquifer. The actual technologies used can be both biological, or physical/chemical
- *Ex situ measures*: measures where groundwater is treated above the ground. The most common kind of ex situ measures are *pump-and-treat* technologies. Here, groundwater is pumped to the surface, treated with biological or physical/chemical purification technologies, and then percolates back through the soil.

In both categories, a range of technologies are available; e.g. the US Federal Remediation Technologies Roundtable identifies 12 in-situ and 11 ex-situ technologies, the majority of which use physical/chemical treatment. The choice of the appropriate technology is highly site-specific. A survey paper of the US EPA (1999) summarises the experience from 28 different remediation sites throughout the US,[20] and, from this experience, identifies six main factors that determine the cost and effectiveness of a remediation technology in practice:

- *Source control factors* – include the method, timing, and success of source controls to mitigate contact of contaminant sources with groundwater
- *Hydrogeologic factors* – relate to the properties of the Aquifer that define contaminant transport and the specifications of the groundwater extraction system, including hydraulic connection with other aquifers, aquifer flow parameters, and influences from surface water bodies
- *Contaminant property factors* – capture the contaminant properties that determine how easily contaminants can be removed from the aquifer, the steps that are required to treat the extracted groundwater, and the complexity of the contaminant mixture
- *Extent of contamination factors* – measure the magnitude of the contaminated groundwater plume, including its area and depth and the concentrations of contaminants
- *Remedial goal factors* – include the regulatory factors that affect the design of a remedial system and the duration through which it operates, such as the aquifer restoration or treatment system performance goals or specification of a particular treatment technology
- *System design and operation factors* – capture the adequacy of the chosen technology to the site-specific requirements, the efforts made to optimise the system, the amount and type of monitoring performed, and the combination of different technologies

Because of the range of factors that determine the appropriateness and effectiveness of a remediation technology, it is not possible to offer general conclusions about the optimal choice or the cost-effectiveness of any particular technology in the context of this study. However, considering the cost-efficiency of restoration technologies in general, it appears that more often than not, groundwater restoration will not be possible in a cost-efficient manner. The US Commission on Geosciences, Environment and Resources takes the view that generally, the restoration of contaminated aquifers should be viewed with much caution. Even where it is technically feasible, it remains a resource-intensive and time-consuming activity. Although new restoration technologies are developed, their effectiveness remains uncertain. As a general result, it will almost always be less expensive to prevent groundwater contamination than to clean it up (CGER 1997).

4.2.2 Containment

Containment comprises all measures used to prevent further spread of a contamination plume. According to the distinction made by the Flemish Public Waste Management Author-

---

[20] A broader overview of the experiences with and the costs of different groundwater remediation technologies in the US can be gained at the homepage of the Federal Remediation Technologies Roundtable, http://costperformance.org/studiescat.cfm

ity, these measures are generally effect-oriented; i.e. their aim is to isolate, limit and control the source of contamination (OVAM 2002). Consequently, containment measures are only applicable in cases of point-source pollution, where the pollution is still limited to a relatively small area, e.g. in the case of contaminated sites. Preventing further spread of the contamination can be achieved by means of *underground dikes* or *slurry walls* that are built into the aquifer to isolate the pollution source. The cost and feasibility of such measures depends largely on the hydrogeological situation and the size of the contamination plume.

One technology that combines elements of containment and restoration are *permeable reactive barriers* installed across the flow path of a contaminant plume. Such barriers contain or create a treatment area that removes contaminants from the groundwater flowing through it. Contaminant removal occurs passively and can be done using both physical/chemical or biological processes.

A different containment option consists of the abstraction and disposal of contaminated water; this method is referred to as *counterpumping*. The Case Study on the potash mining fields in Alsace, France, explains one such case where groundwater with an extremely high salinity is abstracted from a contaminated plume and lead into the Rhine river. One limitation of such measures concerns the disposal: either pollutant concentrations in the abstracted water have to be sufficiently low, or the recipient water body into which the polluted water is pumped has to be sufficiently large, so that negative effects can be excluded.

In general, a similar caveat applies to containment options as it does to restoration: in general, containment of contamination will almost always be more costly than protecting groundwater from contamination. However, the choice for any particular containment option will have to be made depending on local hydrogeological conditions and on the kind of pollution; general recommendations cannot be made. In addition, it should be pointed out that containment can only prevent further spread of contamination, but does not solve the problem at its source. Experiences with contaminated sites that were dealt with under the US Superfund programme have shown that cleaning up the site by treating the contamination would have been more cost-effective than containment: although the costs of treatment are higher initially, it may be the cheaper option in the long run if the running costs are taken into account (Office for Technology Assessment / OTA 1985, cf. ch. 7).

4.2.3 Removal at Discharge Points

As elaborated in chapter 3.3.1, wetlands have the capacity of retaining toxic substances and removing nitrates through sedimentation, uptake in biomass or denitrification. Since many wetlands are located, or can be created, at the discharge points of an aquifer, it is one possibility to utilise this retention and removal potential of wetlands as a means of filtering out groundwater pollution (the technical and economic feasibility of such measures is discussed in Byström (1998) and Söderqvist (1998). This option is not a remediation measure in the strict sense, since groundwater contamination is not corrected in situ – the quality of the aquifer itself is unaffected by this.

A different case arises if constructed wetlands are used to reduce the nitrate content in run-off from fields, as long as the nitrogen volumes are not too high. In these cases, nitrate retention appears as a more cost-effective method than pollution reductions through changed land use practices. This result would be further reinforced if the social values of wetlands (in terms of amenity, support of biodiversity etc.) were included – which they were not in the reviewed literature. However, nitrate removal through constructed wetlands is targeted at surface water, and contributes to groundwater protection only insofar as it may prevent polluted surface water from percolating to an aquifer. It should also be noted that nutrient removal and retention of toxic substances is limited to moderate contamination levels; for highly concentrated or continuous discharges, the wetland itself will be impacted.

4.3 Conclusions on the Instruments for Protection and Remediation

Concerning the comparison of different instruments and strategies in groundwater protection, a general caveat has been raised by Raucher (1983). He points out that the costs of dealing with groundwater contamination depend on which path of action is chosen: a given environmental target can be achieved with a range of measures, and the optimal response always depends on the target. In exceptional cases, it might even be economically rational to suffer the damage from contaminated groundwater, rather than taking action to clean it.

Whether such a solution is preferable depends not only on the extent of contamination and its expansion over time, but also on the current and future uses of the groundwater, including the effects of groundwater pollution on groundwater-dependent ecosystems. In addition, it must be considered that economic rationality is only one of several foundations on which decisions can be based, and that it depends on the currently available knowledge.

4.3.1 Distribution of Costs

Environmental regulation inevitably benefits some groups more than others. While the distribution of gains and losses is irrelevant when overall economic efficiency is considered, it is crucial in determining public support for environmental regulation, a point raised by Söderqvist (1998b). Regulation will only persist if it is perceived as fair and appropriate; to this end gains and losses have to remain in balance. If the hardships from regulation are perceived as unacceptable by one party, it may be appropriate to compensate a part of the losses - ideally with taxes raised from the party which benefits from regulation.

- The beneficiaries of better groundwater protection are above all private households, which depend on groundwater as their drinking water supply (see chapter 6), as well as water supply companies that will face less expenditure for monitoring and purification. Benefits also accrue to firms that depend on clean water supplies, e.g. in the food or beverage industry. A considerable part of the benefits, however, will take the form of more healthy groundwater-dependent ecosystems and the preservation of a valuable resource for future use. These benefits accrue to society in general, and to future generations in particular.
- The costs of groundwater protection will obviously be felt foremost by the regulated sectors, however the extent of the cost depends considerably on the type and design of protection measures. For example, in the case of pollution taxation, the effects depend not only on the level of taxes, but also on the ways that revenue is used. On the other hand, it should be noted that the regulated industries have also benefited from the right to emit freely in the past. Gardner and Young (1988) quote the case of a government-funded pollution abatement programme that has effectively amounted to a massive subsidisation of farmers, to the extent that the benefits to the public from reduced nitrate pollution do not even equal the amount of compensation received by farmers.

The question of how environmental costs should be allocated, especially in the context of water pricing, involves some complex decisions. Two extreme cases can be distinguished, mirroring the distinction drawn by Coase (1960) concerning the distribution of the right to pollute (cf. chapter 2.2):

- If the costs for groundwater purification and treatment, or even the costs of groundwater protection, are borne exclusively by water consumers (i.e. private households and firms), this effectively amounts to implementing the victim-pays-principle. The implicit assumption is that consumers do not have the right to unpolluted groundwater, and must therefore bear the cost of purification. This view is restricted to the function of groundwater as a source of drinking water – the capacity to absorb pollutants is viewed as a free service.
- At the other extreme, if it is regarded as the consumers' categorical right to have access to unpolluted groundwater, then the costs of water purification would have to be borne entirely by the polluters. In this case, the users of drinking water would only have to pay

the costs of abstraction and transporting, as well as some price component to reflect the scarcity of groundwater, and the clean-up cost for the waste water that they emit. The decision would be on the polluters in this case, whether they prefer to reduce pollution and bear the cost of it, or whether they would rather pay the purification costs. An even stronger scenario would make polluters accountable for the costs of restoring a polluted aquifer, where possible; this would amount to a full implementation of the polluter-pays-principle.

Of course neither of the two extreme cases will be implemented in practice. Whereas the victim-pays-principle contradicts the fundamentals of EU environmental policy and law, a full implementation of the polluter-pays principle is also unlikely for a number of reasons:

- *Political reasons*: a full-scale implementation as described above would create a disproportionate pressure on the affected parties (especially farmers) and therefore is not likely to find sufficient political support. This is also supported by Olson's "logic of collective action": the potential losers constitute a relatively small group, with a relatively large stake to lose; it is therefore likely that they will organise themselves and take action against regulation. In contrast, the potential beneficiaries are a much larger group, but with comparatively small individual benefits. Therefore it is unlikely that they will organise themselves to lobby for stricter regulation (Olson 1965)
- *Technical reasons*: as noted above, the restoration of an aquifer to its unpolluted state is often impossible, or connected to prohibitively high costs. To place these costs on the polluter would be questionable from both a social and an economic point of view
- *Legal reasons*: placing the total cost of pollution on the polluter requires complete and reliable information about the emissions of each individual polluter. While this problem can partly be overcome by taxing the polluting substances rather than the emissions, the monitoring requirements would nonetheless be tremendous.

4.3.2 Choice of the Instrument Mix

Bearing in mind the limitations outlined above, the following conclusions can be drawn:

➢ ***Choice between remediation and protection***

In general, prevention of pollution is almost always less costly than clean-up. However, protection is not always possible; in cases of historical or accidental pollution, remediation must be considered.

➢ ***Choice among protection instruments***

Regulative approaches have been the backbone of groundwater protection and will remain a central part of it. Their advantage is that they (theoretically) allow for a more targeted setting of environmental quality goals. However, at the same time there is considerable scope for the extended use of economic instruments in groundwater protection.

➢ ***Taxes***

From a theoretical perspective, an emission-based tax system would be a highly efficient option for groundwater protection; however, due to the associated monitoring requirements, it is technically unattainable. Instead, taxes on inputs for polluting processes offer a second-best alternative. It must be noted that the implementation of taxes to control agricultural diffuse pollution is a political rather than an economic problem.

➢ ***Subsidies***

The proposed reform of agricultural subsidies, not least to strengthen environmental considerations, offers considerable scope for a stronger representation of groundwater protection requirements (issue of cross-compliance).

- ***Cooperative agreements***

Cooperative agreements are a feasible solution in principle, and have been implemented in different European countries, because they offer a cost-efficient alternative to other instruments and are relatively uncontroversial instruments. However, their distributional impact depends on how the right to pollute is regulated in the respective country; the instrument may result in solutions where the victim has to pay for pollution.

- ***Informational measures***

A potential for win-win-solutions exists whereby environmental improvements can be achieved at no direct cost for the regulated parties. The available scope for such no-regrets solutions through improved irrigation, fertiliser or pesticide management must be exploited. Although their effectiveness may be limited, informational measures offer inexpensive and uncontroversial possibilities for improvement.

- ***Choice among remediation technologies***

The costs of remediation depend on the specific technologies used. Technical information about costs and effectiveness of a wide range of technologies is available, however this information is difficult to summarise and structure because the optimal choice of technology depends on the nature of the contamination, on the site-specific hydrogeological conditions and on the urgency of the problem.

5 The Cost of Protection and Remediation

The previous chapter has discussed different instruments available for the protection and remediation of groundwater. This chapter extends this discussion by providing evidence of the costs associated with different measures. In order to facilitate the comparison of results, the presentation of evidence on the costs is structured around the various pollution problems rather than instruments. Chapter 5.1 presents some evidence of the cost of protection measures that reduce diffuse-source pollutions, whereas chapter 5.2 turns to the costs of dealing with point-source pollution.

5.1 Cost of Protection from Diffuse Sources

The following cases offer some evidence of the cost of reducing agricultural diffuse pollution. These cost mainly take the following forms:

For reduced *fertiliser applications*, they consist of

- diminished agricultural productivity, i.e. less crop yield per hectare as well as less intensive farming practices;
- information gathering for better fertiliser management;
- changing to different crops, or to different combinations or rotations of crops;
- switching to alternative land uses, i.e. from tillage to pasture or forestry.

For restrictions on *pesticide use*, costs arise from

- reduced agricultural productivity due to unwanted weed and parasite species;
- information gathering for better pesticide management;
- employing alternative, more costly weed eradication methods.

5.1.1 Costs of Reducing Nitrogen Applications

Johnson et al. (1991) present an economic analysis of the costs of groundwater protection for farmers, supported by empirical evidence from farmlands in the Columbia Basin, Oregon, US. They find that a uniform reduction of nitrogen application leads to a decrease in nitrogen leaching of only 2-4%, depending on the crop, while the yields were reduced between 10 and 22%. Thus, according to their simulation models, a uniform reduction strategy would be highly inefficient from an economic point of view, leading to minimal results at a high cost.

In this context, the findings of Rejesus and Hornbacker (1999) should also be considered (cf. Box 4.8). They found possibilities to reduce nitrate application and groundwater contamination levels significantly at no additional cost, simply by improving irrigation management. They argued that farmers frequently do not take local soil and weather conditions into account when deciding fertiliser application levels. Therefore, in some cases they tend to go beyond the level that maximises their profits, so that reducing fertiliser levels both increases farm profitability and benefits the groundwater.

Gramel and Urban (2001) have researched the cost of reducing nitrate concentrations in the soil for the case of cooperative agreements concluded in the German Bundesland Hesse (cf. chapter 4.1.3 for a detailed discussion). Based on the nitrate reductions in the soil (of 20 to 60 kg N/ha) that were achieved through the agreements, the authors calculate an average cost of 0.29 €/m^3 of abstracted groundwater.[21] Compared to costs of 0.25 to 0.75 €/m^3 for end-of-the-pipe treatment of groundwater used as drinking water, the authors conclude that

[21] The reported costs range from 0.05 to 1.05 Euro/m^3, depending on how much groundwater is abstracted in the respective area; 0.29 Euro/m^3 is a weighted average.

cooperative agreements are a competitive alternative even if the analysis considers drinking water supply only.

5.1.2 Costs of Reducing Pesticide Applications

Archer and Shogren (2001) assess different forms of pesticide taxation and compare them with an outright ban on selected pesticides. Based on the a computer-based yield simulation model, they evaluate the impact of a flat tax, a tax based on the predicted health impact of the pesticide, and different taxes based on the pesticides' relative risk of reaching and damaging the groundwater. Different tax levels are assumed, ranging from US$ 1 - 15 per pound of active ingredient (or some equivalent measure). The income losses that result from the different taxation schemes are generally moderate, in the order of up to 4% of the baseline profits. The costs are highest for the taxation scheme based on the impact on the aquatic environment, and not much lower for a flat tax of 10 or 15 US$ per lb. On the other hand, the cost are lowest with a taxation scheme based on exposure values (combining the risk that herbicides pose to the water supply with their likelihood of reaching the groundwater). For these schemes, the costs are less than 0,25% across all tax levels. The complete ban on selected pesticides ranges somewhere in between, with costs around 1,6% of the baseline profits.

5.1.3 Costs of the Protection of Catchment Areas

Pfaffenberger and Scheele (1990) provide an estimate for the costs of protecting groundwater catchment areas in Germany. They argue that the protection of catchment areas is the most widely used instrument for groundwater protection in Germany. However, the avilable legal instruments for designated protection areas and concomitant usage restrictions are not fully enforced for a variety of reasons: Among others, water suppliers may fear the cumbersome and expensive legal procedures needed to establish and enforce protected areas; the costs of restricting certain economic uses may be prohibitive; and, for this reason, there may be a lack of public support from local municipalities. To circumvent these problems, water suppliers have engaged in buying up land in the catchment areas and using it for forestry. The authors have calculated the hypothetical cost of implementing this approach on a large scale, covering between 10 and 30 per cent of the areas that are now designated as protected catchment areas – which amounts to 7.7 per cent of the surface area of Germany. The calculated costs consist of the land price itself, the necessary investments in reforestation, and the maintenance costs. They estimate the total cost of acquiring the land at € 465 billion (in 1990 prices), with an annual running cost of € 2.5 billion. Due to differing land prices and protection requirements, there is a strong regional variance in costs, however the weighted average cost of this measure is 55 ¢ per ha per year.

Messner et al. (2001) present a case study which evaluates the use conflicts associated with protection zones in the Eastern German region of Torgau. In the study region, almost two thirds of the surface area are designated as protected areas, sometimes placing severe restrictions on economic uses. The authors compare the effects of four alternative policy scenarios over a period of 30 years; the scenarios differ in terms of whether the protected areas are decreased or remain unchanged, and whether gravel production is increased or remains the same.

For the calculation of the monetary costs of this approach, Messner et al. include the agro-economic return per hectare, the foregone income in the gravel industry, the economic and environmental costs of gravel transports from other regions, and investments into special protection structures - e.g. for infrastructure, roads and underground storage tanks. From the economic analysis it becomes clear that the costs are dominated by the foregone returns from gravel production. However, these work in the opposite direction from what could be expected: the scenarios which do not foresee an increase in gravel production are more beneficial economically than those where more pits are opened; this is due to the fact that there is already an oversupply of gravel in the region. The total (undiscounted) cost from the latter scenarios ranged between € 3,0 bn and 0 (for the reference scenario), while the cost of the scenarios without additional gravel production ranged from € -9,1 to -10,9 bn. Here, the negative costs indicate that the additional profits from gravel production if *no* other pits are opened outweigh the other cost factors by far. Net of the gravel production effect, the costs of the protected area are € 1,8 bn for both scenarios where there is no change in protected areas.

5.2 Cost of Protection from Point Sources

The US EPA (1999) offers a survey of 28 groundwater remediation projects throughout the US, which differed both in the technologies used, in the remediation goals, and in the spatial extent of the contamination. In interpreting the results, it should be noted that the projects were not chosen to build a representative sample, therefore it is unclear to what extent the findings can be generalised.

Of the 28 projects, 26 were realised using pump-and-treat technologies (cf. chapter 4.2.1). For these, the costs for the median initial investment are reported at $1.9 million, with a median annual operating cost of $190,000. Bearing in mind that the pump-and-treat systems used had very different capacities, ranging from 6,400 to 2.08 million m^3 per year, the authors also report unit costs per m^3 of treated groundwater per year. On average, these amounted to capital costs of $25 / m^3, as well as $4.75 of average annual operating cost per m^3 treated.

In addition, the survey reports average estimates for three systems using permeable reactive barriers (cf. chapter 4.2.2), a technology combining elements of restoration and containment. For these systems, median capital cost were estimated at $ 500,000, and average operating cost at $ 85,000 annually. Per unit costs are reported as $ 25 of capital cost per m^3 of groundwater treated annually, and $ 4.75 of average annual operating cost per m^3 treated.

Box 5.1: The Cost of Groundwater Remediation in Belgium and the UK

Ecolas (2002) gives an overview of the cost of groundwater remediation measures in three different sectors in Flanders. The report for the Flemish Public Waste Management Authority (OVAM) considers all firms for which a soil investigation procedure is mandatory under Flemish law. On the basis of past experiences, the expected costs for cleanup measures are extrapolated into the future.

The average cost of remedial activities in *all* sectors is estimated to be € 600,000 per site, whereby 60% of all remediation projects cost less than € 100,000. This means that the average costs for the clean-up of contaminated sites are largely determined by the few most expensive projects: the bulk of the total costs is caused by the top 3,5% of all projects. This category comprises 14 project with an average cost of more than € 12 million each.

The study focuses on three sectors in particular: the garage sector, the dry cleaning sector, and metal processing firms.

- In the garage sector, the average cost of remediation projects is € 90,000, with 80% of all projects costing below € 100,000.
- In the dry cleaning sector, empirical results are scarce. Therefore the study calculates the hypothetical clean-up cost for a site contaminated by chlorinated hydrocarbons (VOX); this leads to an estimated cost of up to € 200,000 per site.
- In the metal processing sector, the average cost per site is estimated to be € 550,000, whereby a third of all firms face costs smaller than € 50,000.

Some insights can also be derived from the British Environment Agency's decision guideline for choosing the optimal remediation approach and technology (EA Technical Report P2-078/TR, Environment Agency 2002). The report aims to present a decision making framework to the clean-up of an exemplary, hypothetical site.

The presented decision making mechanism has been designed to aid the selection of an economically optimal remediation approach. In order to determine the optimal level, decisions have to be made on different hierarchical levels. On the most fundamental level, the policy objective is taken as a given; this relates to the overall decision on what to protect, what to remediate and what to sacrifice when dealing with soil and groundwater contamination. The following parameters have to be determined in order to find the optimal strategy:

1. remedial/risk management objective setting;
2. remedial approach analysis;
3. remedial technology selection.

The *remedial objective* can be the removal, reduction or isolation of contaminant, or the protection of a receptor. The possible *remedial approaches* are to remove the contaminant, to intercept the link between contamination and receptor, or to protect the receptor. On the most applied level, the *remedial technology selection* aims to determine the lowest-cost option to achieve a certain given objective.

This framework is then applied to hypothetical hybrid site with multiple contaminations. The stated aim of this exercise is to set a cost-efficient remedial objective for the site, taking into account the complex contamination history and the relations between multiple objectives. The costs for different remediation measures at the site are reported ranging from GB£ 0.5m, for a partial excavation of contaminated superficial deposits, to GB£ 1.95m for collection and treatment of discharges from the contaminated area to an adjacent river. The reported costs are discounted present values for a treatment period of up to 20 years.

The framework provides a good example of combining costs arising at different points in time, and monetised as well as non-monetised benefits of different remediation measures, in order to choose an optimal remediation objective and technology. It also shows that the calculation of net benefits for any technologically feasible option is not the only aspect to be considered in the choice of an optimal approach, but that other factors such as likelihood of success or the degree of objective satisfaction must also be considered. From this, the authors conclude that the role of an economic appraisal is to “provide information on key factors to aid decision-makers select between options. It is unlikely ever to be sufficiently precise to provide ‘the answer’ and should be used as one tool in the wider decision-making process.”

6 The Benefits of Groundwater Protection

Groundwater provides manifold services – of which the provision of clean drinking water is only one.[22] Only few of these services are traded on the market; therefore, there is no market price for most of them. However, in order to determine the social and economic benefits of groundwater protection, a monetary value has to be put on the different services, and it must be assessed in monetary terms how the services are affected by pollution – which will facilitate a judgement on whether the costs of groundwater protection are warranted by the benefits.

Because of the variety of services, there is also a variety of mechanisms that can be used to assess the value of groundwater. One thing, however, is common to all of them: the benefits of groundwater protection can be seen as avoided damage costs, in one form or another. Rather than direct economic gain, the benefits take the form of fewer damages, fewer risks and anxiety, or less defensive expenditures for groundwater users. The benefits are therefore only partly economic, but also include increased well-being, food security and health. In this sense, assessments of the *damage* from *increased* groundwater pollution can be seen as assessments of the *benefits* from *reduced* pollution.

This chapter discusses the different approaches that can be used to put a monetary value on the services that groundwater provides. This is done in five subsections. Chapter 6.1 explains the economic theory behind the assignment of monetary values to a non-traded good such as groundwater. It also presents the different approaches used to measure such values, bearing in mind that different kinds of damage require different valuation methods.

The second and third parts present evidence of the benefits of groundwater protection, derived from two different types of estimation procedures: chapter 6.2 presents various estimates of the benefits from *avoided damage costs* that go along with a pollution reduction. The methods used include measuring the costs of *averting behaviour* by households in order to avoid polluted groundwater, the *treatment costs* for purification of polluted groundwater intended for use as drinking water, and the assessment of *illness costs* caused by polluted groundwater. Chapter 6.3 discusses examples of *contingent valuation* studies, in which households are asked to state whether and how much they are prepared to pay for measures that improve groundwater quality.

Chapter 6.4 then turns to the indirect value of groundwater, which is transmitted through its effect on groundwater-dependent ecosystems and surface water flows; these effects are also referred to as *ecosystem benefits*. Since these aspects have not been the centre of much research, the assessment of ecosystem benefits will be restricted to a general discussion. Finally, chapter 6.5 offers some conclusions on the different valuation methods and their merits, and on the results derived from them.

6.1 The Value of Groundwater

This section explains the standard economic approach to valuing a good which is – at least partly – untraded. It first discusses the application of valuation in general to the case of groundwater, and the difficulties of this approach (6.1.1). It then turns to the fundamental distinction between different parts that constitute the total economic value of groundwater, which is the sum of its *use values* (6.1.2) and its *non-use values* (6.1.3). Along with the discussion of use values, the different estimation procedures used to assess the benefit of improved groundwater quality are introduced. In addition, the *indirect value* of groundwater, which is transmitted through its contribution to sustaining other ecosystems, will be ad-

[22] Note that *services*, in this context, refer to all the functions provided by groundwater that are in someway useful or important for humans, both directly and indirectly. It includes the supply of drinking and irrigation water, but also the sustenance of surface water flows or the reassuring knowledge that groundwater supplies will be there in the future. Services do not need to be marketed in order to be relevant.

dressed in 6.1.4. Chapter 6.1.5 concludes the theoretical introduction with a short overview of alternative approaches for categorising the value of groundwater.

6.1.1 Valuing Groundwater: Theory, Relevance and Limitations

By way of introduction, it is helpful first to asses the relevance and the difficulties of placing a value on groundwater and groundwater quality. Valuing groundwater provides for the capturing of benefits of groundwater protection in monetary terms: it provides part of the answer whether the costs of protection measures are justified by their benefits.[23] As the cost are readily available in monetary terms, it is helpful to put a value on unpolluted groundwater in order to make costs and benefits comparable.

Box 6.1: The difficulties of Pricing Groundwater as a Non-Marketed Good

Putting a value on a good for which there is no market price is a difficult exercise: the price of a good signals how much it is worth to the average consumer, i.e. how much consumption of other goods the consumer is prepared to give up for it. For any normal good, the price of a good is determined through the market. For groundwater this is not the case, since there is no market for it in general. Only selected services of groundwater can be traded – for example, there is a price for drinking water. As a rough proxy, it would be possible to estimate the value of groundwater based on the average price of drinking water. However, this approach would lead to a gross understatement of the full value of groundwater for three reasons:

- It ignores the *multitude of services* provided by groundwater. These range from irrigation water supply to the sustenance of surface water flows. The provision of drinking water is only one of these services – which is very important economically, but which affects only a small portion of all groundwater in many countries. A valuation of groundwater also has to take the other services into account.
- It ignores *scarcity considerations*. As any price, the price of drinking water is determined at the margin: this means that it optimally reflects the cost of providing one additional unit of a good. Fortunately, in most European regions there is usually a sufficient supply of clean drinking water, or alternative sources are nearby, so that additional water can be supplied at little extra cost. However, this marginal price cannot be used to calculate the total value of all groundwater – because, for the whole of European groundwater, there is no easily available alternative source, from which all water could be supplied at the same cost.
- It ignores the *market conditions* on the water market. Although liberalisation of the water market is progressing throughout the EU, many aspects of water supply are still regulated and will remain so for various reasons – for example, exclusive private property rights for groundwater are ruled out legally in most countries, and would certainly not be desirable (see below). Therefore, the price for drinking water is not representative of the 'true' market price that would avail under completely liberalised market conditions.

From an economic perspective, it must be noted that groundwater has the characteristics of a public good in many respects (cf. chapter 2.3). For these public good functions (e.g. the sustenance of groundwater-dependent ecosystems, or the use as a water reserve in times of draught) there cannot be a market price, because there is no rivalry in consumption or excludability from use. A different point is that there are no private property rights for groundwater as such, since groundwater is commonly regarded as a good worthy of special protection out of public interest. Therefore, land ownership is either legally separated from the ownership of the groundwater below, or is subject to extensive use restrictions. For these reasons, a market mechanism that would reflect the supply and demand for clean groundwater is not even a theoretical possibility.

---

23 There are also some efficiency tests that do without monetary benefit estimates, such as cost-efficiency analyses or cost comparisons. These will be explained in more detail in chapter 7.1.

It is clear that there is no readily available price of groundwater, neither can its value be simply derived from the price of other, marketed goods. The challenge is therefore to find a reliable way of calculating the value of groundwater, which allows a comparison with the costs incurred to protect it. In environmental economics, the relevant measure for the valuation of a non-marketed environmental resource is its *total economic value* (TEV).

The total economic value is calculated as the sum of a resource's *use value* and its *non-use value*.[24] In the case of groundwater, the use value measures all benefits that can be derived from putting the groundwater to a specific use – the specific application will be explained in chapter 6.1.2 below. By contrast, groundwater can also be valuable to someone even if there is no actual intention of using it; this part of its value is consequently referred to as non-use value. Different kinds of non-use values that can be distinguished in the case of groundwater are presented in chapter 6.1.3.

In addition, in the case of groundwater, it is also necessary to consider a class of indirect benefits referred to as *ecosystem benefits*. One important service of groundwater is to sustain surface water flows and groundwater-dependent ecosystems. These surface water bodies and ecosystems themselves have an economic value – a part of which can be attributed to groundwater, in the sense that the value of the resources would be diminished if groundwater discharges were reduced, or their quality deteriorated. These indirect effects are not usually included in the total economic value of groundwater. As the Commission on Geosciences, Environment and Resources acknowledges, a possible reason for this is that ecosystem benefits have only become a concern in recent years as the interaction between different aquatic ecosystems has become better understood (CGER 1997). These issues will be discussed more extensively in chapter 6.1.4.

For a full economic assessment of groundwater protection, it would not be sufficient to put a value on clean groundwater as such, regarding groundwater quality as a binary variable that is either good or bad. In addition, it is necessary to assess how this value changes for different quality levels. This information, in the form of an economic dose-response function, would be of great help in determining optimal protection levels. Unfortunately, such information is very difficult to find. Apart from this, some limitations concern the following fields:

- *Transferability of results*. Estimates of the value of groundwater can only be obtained through local case studies, and are therefore always site-specific. Although the long-term goal is to arrive at a comprehensive valuation function, which would allow transferring results from case studies to other areas, this is not possible with the knowledge base that is available today.
- *Completeness of results*. As argued above, even the *total* economic value may not be a complete measure of the benefits from protection if it fails to acknowledge indirect ecosystem benefits. Since the whole field of groundwater valuation is still fairly young and has not been researched exhaustively, it remains doubtful whether it is possible to capture the entire value of groundwater.
- *Hypothetical nature of the results*. The practical benefits of groundwater protection are real and tangible. However, value estimates derived with economic measures are at least partly constructed. Therefore, as the knowledge of valuation methods develops, changes in the values themselves are also possible. This is especially relevant for *contingent valuation* studies (cf. chapter 6.3).
- *Concrete nature of the costs*. In comparing costs and benefits, a structural problem is that costs are manifest for those who have to bear them. Therefore, in most cases, these

[24] Some alternative interpretations can be found in the economic literature on groundwater, e.g. the distinction between *extractive* and *in situ* values of groundwater. These alternative, but roughly equivalent categorisations will be summarised very briefly in section 6.1.5.

costs are also immediately available in monetary terms. By contrast, the benefits have to be calculated with sophisticated and costly methods (cf. chapter 7.1).

6.1.2 Use Values of Groundwater

The concept of use values captures those impacts on groundwater that lead to concrete adverse effects. Such damages occur because consumers are forced to change to more costly sources of drinking water in order to avoid groundwater contamination, or because they are affected through negative health effects or through increased fear and anxiety.

Whenever private or commercial groundwater users are affected by pollution, it is necessary to assess in monetary terms the damage arising from polluted groundwater, in order to include pollution into an economic decision-making process. By the same rationale, the benefits associated with a pollution reduction can be viewed as the reduced costs of groundwater contamination. Ideally, this takes the form of a *damage function*, which relates the level of pollutants emitted to the total economic costs. To assess the economic impact of pollution, it is also necessary to find a baseline condition which measures the quality of groundwater in the absence of any damaging human interference. Since the natural background concentrations of some pollutants even in pristine groundwater can be significant, the task of defining this baseline is not as easy as it seems. In addition, the dynamic character of the groundwater contamination must be taken into account: ideally, the baseline condition should give the state of groundwater at any point in time, taking into account expected changes in quantity and quality.

If there is a manifest damage from pollution, costs are relatively straightforward to measure: these include reduced crop yields associated with polluted irrigation water, falling output or damaged reputation for firms that rely on pure water supply, or the cost of treating illnesses that can be traced back to polluted drinking water. Both of these could be quantified comparatively easily based on the market prices of the affected goods.[25] However the reliability of such estimates is limited by some factors:

- Establishing the link between water quality / pollution and the damage in physical terms may be difficult. For instance, the marginal impact of water quality on health is difficult to assess because diseases are provoked by many factors other than water quality – such as lifestyles, alimentation or stress.
- If there were indeed a significant health threat from contaminated groundwater, consumers would certainly stop using it as a source of drinking water. Similar considerations would apply if polluted irrigation water led to a health risk from crops.
- For the cost of reduced agricultural productivity, similar caveats apply: only in some cases, such as boron and NaCl pollution, groundwater pollution will impact plant growth. A more likely scenario is that crops may be harder to sell if there is concern about the health risks of contaminated irrigation water.

In reaction to the fact that consumers and water suppliers incur costs in order to reduce their exposure to contaminated groundwater, a different approach is to see these defensive expenditures as an indirect valuation of groundwater. In this way, the *avoided damage cost approach* does not look at the costs of an actual damage, but rather at the expenditures that are necessary to prevent such damage from occurring.

Three different kinds of avoided damage costs will be elaborated below (cf. chapter 6.2): the *averting behaviour approach* approximates the cost of polluted groundwater based on

[25] For the valuation of impacts on agricultural productivity, Johnson et al. (1990) point out a particular problem. Since the agricultural sector is subsidised in all industrialised countries, it is difficult to determine reliable market prices (net of state interventions). However, such 'true' market prices would be needed to reliably assess the cost of reduced productivity as a consequence of polluted groundwater. Using current prices is likely to overstate the damage caused by polluted groundwater.

what consumers pay for using bottled water instead of tap water, whereas the *treatment costs* measure how much water suppliers invest in purification devices. Finally, the *cost-of-illness approach* measures the cost of sicknesses induced by contaminated groundwater.[26]

Box 6.2: Other Examples of Revealed Preference Approaches

The avoided damage cost approach is an example of a *revealed preference* approach, which deduces consumers' valuation for cleaner groundwater from their behaviour. Apart from avoided damage cost, the most prominent methods are *hedonic pricing* and *travel cost* approaches. Whereas some applications of the avoided damage cost approach are discussed in chapter 6.2, hedonic pricing and the travel cost method are not applicable to groundwater protection as such, and are therefore of indirect relevance only.

- Hedonic pricing aims at deducing a resource's value from its effects on the values of other, marketed goods. One typical example is the effect of noise from a nearby airport on house prices. However, in most cases, groundwater quality will not have a large enough impact to lead to measurable changes in house prices, or in the prices for agricultural land. To our knowledge, there is no application of the hedonic pricing method to estimate the value of groundwater.[27]
- The travel cost method, by contrast, is based on the idea that the value of a natural resource can be deducted from the time and money consumers invest in travelling to the resource. Again, it is unlikely that such an effect would be measurable for groundwater directly since groundwater as such obviously has neither recreational value, nor is it of tourist interest.[28] Consequently, to our knowledge no such studies has been published.

One aspect that complicates the estimation of the benefits of clean groundwater is the variety of economic uses for (ground-)water. For different uses, the requirements in terms of purity will also differ, according to whether groundwater is used as drinking water, as input in the food and beverage industry, as drinking water for lifestock, for irrigation, for industrial processes or simply as cooling water matters for the required quality level. Hence a damage function, which relates pollution to economic impact, will be *discontinuous*. At certain threshold levels, it will become necessary to discontinue the use of groundwater for a specific purpose. The activity for which the water was previously used will then either cease, or will rely on water from other sources. Some forms of pollution may also affect certain uses only. Nitrate and pesticide contamination for example are highly relevant for drinking water – but for irrigation purposes, they are irrelevant, except in the case of organic farming.

Groundwater contamination does not only lead to costs if damage has already occurred: if people feel upset about the threat of drinking potentially contaminated water, this can also be seen as an economic cost. Therefore, if consumers state their willingness to pay some amount of money for safer drinking water or for cleaner groundwater in general, this gives an indication that the current situation is not optimal. This idea is the basis of *contingent valuation* analyses (the valuation is contingent on the proposed improvement in groundwater quality), also referred to as *stated preference* techniques.

A number of contingent valuation analyses have estimated consumers' willingness to pay for a cleaner groundwater by means of questionnaires or interviews; these are discussed in chapter 6.3. The contingent valuation approach is particularly relevant for groundwater valuation because it is the only technique that permits the capture of non-use values as well (see below). Also, it is popular among economists, because it mirrors the ideal of a free mar-

---

26 Strictly speaking, treatment costs and costs of illness do not belong are not revealed preference methods, because they do not reflect individual's decisions about clean groundwater; they are subsumed here to avoid further subcategorisations.

27 (Cf. results of the forhcoming Drastrup case study) land-use-change in the catchment area and the associated amenity value have led to increases in house prices in adjacent areas. However, this is not a consequence, but a side-effect of groundwater protection.

28 This is different if indirect benefits of groundwater are considered as well, such as a wetland that depends on (clean) groundwater. In this case part of the recreational value of a wetland can be attributed to groundwater (cf. section 6.1.4).

ket where sovereign consumers can decide how much they value one good in relation to others.

6.1.3 Non-Use Values of Groundwater

A different category of values to be considered in the economic valuation of groundwater is that of non-use-values. In the cases mentioned above, costs arose for users of groundwater because their use (as drinking water, irrigation water etc.) was impacted by pollution in one way or the other. However, there is also the possibility that people feel troubled by the contamination of groundwater which they are not using at present - be it because they intended to use it in the future, or be it because they feel it should be preserved in a pristine state. This notion is captured in the category of non-use values. If groundwater is to be conserved because there is probability that it will be used at a later point in time, this is referred to as its *option* value. If there is a demand to preserve the groundwater in its natural state without any intention of using it whatsoever, this is referred to as its *existence* value.[29] Two other, related types of non-use values are the *bequest value* and the *altruistic value*. Both concepts capture the value that people put on leaving the groundwater resources intact for the use by others - be it other people living today (altruistic value), or future generations (bequest value). Obviously, in many cases the different types of non-use values will not be strictly separable.

Unfortunately, the non-use values of groundwater protection are even more difficult to quantify than the use values. This is because they are not linked to any tradable goods; quantification is only possible through contingent valuation studies (willingness-to-pay analyses, see above). Because these surveys can be tedious, and because of doubts about the robustness of such estimates, non-use values are frequently emitted from the economic values of groundwater altogether. Instead, it is then assumed that the calculated use-values are only the lower bound of the true value of groundwater (see e.g. Raucher 1983).

6.1.4 The Indirect Value of Groundwater: Ecosystem Benefits

The use- and non-use values described above are derived from actual or potential human uses of groundwater: even altruistic and bequest values relate to human uses, albeit by others than those who express their valuation. However, in addition to use- and non-use values, groundwater also has indirect benefits, which stem from groundwater discharges to surface water bodies and groundwater-dependent ecosystems. These effects are commonly referred to as *ecosystem benefits*.

The economic relevance of ecosystem benefits arises from the fact that surface water bodies and groundwater-dependent ecosystems themselves have an economic value. Part of this value can be attributed to groundwater, because the value of groundwater-dependent resources would decline if groundwater discharges were reduced, or if their quality deteriorated. However, the ecosystem benefits of groundwater are even less researched than its direct values, possibly because the interaction between different aquatic ecosystems have only begun to be better understood in recent years (CGER 1997).

A second reason for the lack of reliable data on ecosystem benefits is the enormous measurement effort they require. Clearly, the calculation of ecosystem benefits depends on the value of the affected resources. However, the value of groundwater-dependent ecosystems and surface water flows itself also has to be calculated indirectly, using similar procedures as those described above for groundwater. For example, it would be necessary to calculate the use- and non-use values of a groundwater-dependent wetland, and then estimate how these values are affected by groundwater contamination. To our knowledge, no such calculation has been undertaken; some preliminary estimates are discussed in chapter 6.4.

---

[29] In a way, the existence value can be seen as the economic equivalent of natural resources' intrinsic value, whereas the different use-values and the option value are - at least potential - instrumental values.

An entirely different approach to valuing ecosystem benefits is to argue that they should be seen as part of the non-use value of groundwater.[30] The reasoning is that discharge of groundwater to rivers, lakes and wetlands does not constitute a direct human use of groundwater. It can therefore be maintained that such effects should be reflected for instance in the existence value of groundwater; in this understanding the existence value would not only express the value placed on groundwater as such, but also the value put on groundwater as part of the water cycle. From this perspective, it would be possible then to assess the ecosystem benefits along with other non-use values by means of contingent valuation analyses (cf. 6.3). However, such considerations are neither necessarily included in the non-use values, nor are they likely to be familiar to a majority of respondents – after all, the interactions between ground- and surface waters may not even be fully understood by scientific experts.

6.1.5 Different Categorisations of Groundwater Values

The following table - building on Bergstrom et al. (2001) - gives an overview of the different values that are part of the total economic value of groundwater. The categorisation presented in this table will be used in the following chapters.

Table 1

Overview of the Different Values and Services of Groundwater

	Groundwater service	Cost of pollution	Measurement method
Use values	Drinking water	Health impact and health risks	Avoided damage / Averting behaviour Hedonic pricing Stated preference / Contingent valuation
	Irrigation water	Change in crop values Change in production costs Change in crop patterns	Market prices for crops Cost function estimates Market prices for different crops
	Industrial water use	Change in production costs	Cost function estimates / Factor income method
	Recreational	Amenity loss Fishing / Hunting / Plant gathering	Travel cost method Stated preference Hedonic pricing Travel cost method Stated preference
Non-use Values	(Uncertain) Existence Use by others Use by future generations	Option value Existence value Altruistic value Bequest value	Stated preference Stated preference Stated preference Stated preference
Indirect / Ecosystem Values	Discharge to Ecosystems	Amenity Change in crop values Fishing / Hunting / Plant gathering Biodiversity maintenance	Stated preference Travel cost method Hedonic pricing Market prices for crops Travel cost method Stated preference Stated preference Replacement cost
	Discharge to Rivers and Lakes	Amenity Fishing / Hunting / Plant gathering Biodiversity maintenance	Stated preference Travel cost method Stated preference Stated preference Replacement cost

[30] A comparable approach is put forward by the Commission on Geosciences, Environment and Resources: they subsume the ecological values of groundwater under the category of *in-situ values*, which is roughly comparable to the category of non-use values (CGER 1997) (cf. section 6.1.5 for a summary of the CGER approach).

Box 6.3: Alternative Categorisations of the Value of Groundwater

In addition to the categorisation that will be used here, some different categorisations have been put forward by different authors. For example, CGER (1997) distinguish between *extractive* and *in situ* values of groundwater. In this framework,

- the *extractive value* captures all functions for which groundwater has to be abstracted; it is identical to the use values described above, except for the recreational values.
- *in situ values* are those functions that groundwater provides irrespective of human abstractions. These include the non-use values mentioned above, as well as the recreational use values. In addition to this, it also includes the buffer value, relating to the value of groundwater as a reserve water supply in times of draught; this can be seen as a modified option value. The largest difference is the inclusion of the ecological values, which measure the function of groundwater in supporting dependent ecosystems (cf. chapter 6.1.3 and 6.4)

Abdalla (1994) offers a different approach, which is not structured by the values, but around the types of damage from groundwater contamination. He distinguishes five categories:

- *effects on human health*; these are likely to be chronic effects due to continued exposure to low levels of contamination;
- *increased fear and anxiety* in consumers can be a consequence of groundwater contamination – these will be higher the more uncertain the extent and the effects of contamination are;
- *avoidance costs* incurred by households to avoid using contaminated groundwater, e.g. by using bottled water or by installing purification devices;
- the *ecosystem effects* capture the effect of groundwater contamination on dependent ecosystems, such as wetlands or surface waters; these will be addressed in a separate chapter below (6.4); and
- *non-use-values*, which are defined as above.

6.1.6 The Temporal Dimension of Groundwater Valuation

The natural self-cleaning potential of groundwater aquifers is usually very limited. Therefore the effects of groundwater pollution occurring today will be felt for a long time in the future. However, the different valuation procedures – whether based on use- or on non-use values or both – usually assess the valuation of groundwater only for a short period. In order to decide on an optimal policy, it is necessary to compare its cost with the benefits for the entire planning period, which can easily extend to several decades. Hence, to find the total economic value of an activity, it is necessary to extrapolate the estimated benefits into the future and consequently to sum them up. Should further costs of protection measures be expected in the future, the same must be done for the costs as well (Bergstrom et al. 1996).

In order to compare payments that are made at different points in time, they have to be *discounted*: a payment received today is worth much less than a payment received in 50 years time, therefore the value of future benefits has to be reduced by some factor. This factor is given by the discount rate; as a measure of time preference the discount rate states how much consumers value an increase of groundwater quality in the future against costs that have to be borne today. There is some debate on which discount rate should be used for environmental resources. Especially if the existence and bequest values of groundwater are significant, this should be mirrored in a low discount rate – thus giving more weight to the value of clean groundwater in the future.[31] A discount rate that is commonly used for groundwater protection measures in the US is 3% (ASTSWMO 1998).

The practical consequences of discounting costs and benefits can be considerable; in particular, there may be cases where economic rationality suggests suffering the damage from contaminated ground water, rather than taking action to clean it. This is because the cost of taking action has to be paid now, while the benefits may only materialise far in the future. This situation is reinforced if the contamination is relatively slow to spread over time, or if future use of the groundwater is not foreseeable. On the other hand, as Raucher (1983) remarks, the total benefits of protecting groundwater depend heavily on the time horizon of the calculation: protection measures taken now can deliver social benefits for any amount of time, even if these benefits are discounted; by contrast, if the time horizon is limited to a relatively short period, this attaches more weight to the costs of protection, which are immediate.

6.2 Benefits Estimated as Avoided Damage Costs

While chapter 6.1 discussed the theory behind groundwater valuation, the following chapters will present and discuss empirical evidence from different studies that have been conducted on the valuation of groundwater. The studies will be ordered by the kind of methodology employed – chapter 6.2 presents estimates of avoided damage cost as a direct estimation procedure, whereas chapter 6.3 contains estimates of willingness to pay as an indirect valuation method. Generally it should be noted that only those methods will be discussed for which empirical evidence could be found.

A direct approach of measuring the economic value of groundwater quality is to consider the costs that users have to bear if groundwater quality deteriorates. The underlying idea is that these costs would no longer have to be paid if groundwater quality could be restored: in this sense, the benefits of groundwater protection take the form of *avoided damage costs*. Once again, there are different ways how these costs can be measured. Three of them will be discussed in the following: the averting behaviour approach, the avoided treatment costs, and the cost-of-illness approach. Note that all three approaches are limited to assessing the use-value of drinking water; the averting behaviour and cost-of-illness approaches exclusively focus on drinking water.

6.2.1 Averting Behaviour Approach

The averting behaviour approach confers the valuation for clean groundwater indirectly from the cost that consumers incur in order to avoid contaminated groundwater. Examples of averting behaviour are switching from tap to bottled water, or installing purification devices. One restriction is that the approach only considers health effects of groundwater pollution, and is therefore limited to valuing groundwater which is used as drinking water. Theoretically, it could be extended to health risks from crops irrigated with contaminated water, however, this effect has not been measured. By engaging in averting behaviour, consumers reveal indirectly their valuation for cleaner water - therefore methods based on averting action are also referred to as revealed preference methods. Assuming that consumers would change back to their old behaviour if groundwater quality is improved, the cost of averting action is also a measure of the benefits of groundwater improvement. If, for example, a majority of consumers buy mineral water, not because they dislike tap water but because they prefer the taste of bottled water, then demand for bottled water would be a poor indication of the cost of groundwater contamination.

---

31 This takes up an objection that is frequently raised against discounting of future benefits: with any positive discount rate, the future value of a resource eventually approaches zero. For example, with a discount rate of 3%, a benefit of 100 € 50 years from now equals a present value of just 22 €; and an equally high benefit 200 years from now equals only 27 cent in present terms.

Box 6.4: Averting Behaviour to Avoid Nitrate Contamination in Quebec

One application of the averting behaviour approach was conducted in different regions in the Canadian province of Quebec. Traoré et al. (1999) interviewed 2.333 households in four districts, which were all prone to excessive nitrate concentrations in their groundwater. The survey asked for the use of bottled water or other substitute foods or beverages, as well as whether households had invested in water treatment devices. In three of the four districts, around a third of the respondents used ground water from a private well as their main source of drinking water, whereas in the fourth district the share was at 98%.

Slightly less than half of the respondents reported dissatisfaction with their level of groundwater quality (46%). Of these, almost all (91%) used bottled water for drinking water; a large majority (72%) also boiled the tap water before they would use it, and more than half (56%) of the respondents had installed home water treatment devices. The annual costs households incur for such measures ranged from 156 to 226 CAN$ (about 100 to 145 €).[32] In comparison, the district which relied almost exclusively on private wells for drinking water reported both a higher relative dissatisfaction, and a higher average expenditure on averting measures. When looking at different household characteristics, they found that households with children, households with high exposure levels, and those that considered themselves as environmentally minded reported higher expenditures. The gender of the head of household did not have a significant effect, nor did the educational level.

Abdalla (1994) summarises different studies of averting behaviour and the expenditure consumers incur. He cites a study by Abdalla (1990) on two communities in Pennsylvania that were both affected by groundwater contamination incidents. Whereas in one community awareness of the contamination was almost complete at 96%, of which three quarters engaged in averting activities at an annual average cost of US$ 252, these figures are much lower in the other community. There, 43% of the population were aware of the contamination, and less than half of those incurred averting expenses at an average rate of US$ 123 per year. In a different study, Powell (1991) researched expenditure for bottled water in 15 communities in Massachusetts, New York and Pennsylvania. He finds that those households that live in areas with contaminated groundwater, and are aware of it, spend an average of US$ 32 per year on bottled water, which is four times higher than the average in uncontaminated areas. In addition, these households reported an average willingness to pay of US$ 82 for increased water supply protection. Hence, in this case, averting behaviour clearly mirrors only a small part of the total valuation for cleaner groundwater.

[32] These figures are comparatively low: Traoré also reports estimates from other, comparable studies conducted in the US, which estimated annual expenses ranging from 252 US$ in Central Pennsylvania to up to 2100 US$ in parts of West Virginia or Southeastern Pennsylvania.

The following table from a summary paper by Whitehead and Van Houtven (1997) gives an overview of US studies on different forms of averting behaviour:

Table 2

Averting Behaviour Studies of Safe Drinking Water

Study	Location	Nature and Duration of Episode	Averting Behaviours (a)	Sample Size	Costs (b)
Harrington, Krupnick, and Spofford (1989)	Luzerne County, Pennsylvania	Giardiasis outbreak (12/83 - 9/84)	1, 2, 3	50	$153 -483
Abdalla (1990)	College Township, Pennsylvania	Detection of percholoroethylene in wells (6/87 - 12/87)	1, 2, 3, 4	1012	$26-32
Abdalla, Roach, and Epp (1992)	Perkasie, Pennsylvania	Detection of trichloroethylene in wells (6/88 - 12/89)	1, 2, 3, 4	761	$16 - 35
Collins and Steinback (1993)	Rural West Virginia	Bacterial, Mineral and Organics detected in drinking water supplies (1/87 - 12/89) (c)	1, 2, 3, 4, 5	291,151	$32-36 (d)
Laughland, et al. (1993)	Milesburg, Pennsylvania	Giardia detected in (surface) drinking water supplies (1/89 - 4/89)	1, 2, 3	226	$16-42

(a) 1=hauling safe water, 2=boiling water, 3=purchasing bottled water, 4=installation of home water treatment system, 5=clean or repair water system

(b) Monthly averting costs are adjusted to the monthly household level using 4.3 weeks per month and 30 days per month in 1996 dollars

(c) Dates of water tests for nonpublic water systems, duration of episodes varied by household.

(d) Bacterial - mineral contaminants.

6.2.2 Avoided Treatment Costs

One extension of the averting behaviour approach is to consider the *cost of groundwater purification* for use as drinking water. In most parts of Europe, it is uncommon to use private wells as a source of drinking water supply. Drinking water supply from public networks underlies strict quality controls and, in some cases, receives extensive treatment to guarantee constant quality. Therefore averting behaviour, as discussed in the previous section, would not make sense for consumers connected to the public water supply, unless they have reason to distrust the reliability of the monitoring system. Nonetheless, the costs for groundwater purification have to be borne: in this case, they arise as treatment costs for water supply companies, who will pass them on to the consumers. Analogous to the averting behaviour measures, these purification costs are also costs that could be saved if groundwater contamination was reduced.[33] An extension of the avoided treatment costs are the costs incurred by moving to alternative sources of water supply. This can mean switching to another borehole that draws from the same aquifer, but is located further from a polluted plume; or it can imply importing water from other regions.

Hofreither and Sinabell (1996) offer a calculation of drinking water purification costs for all of Austria. They estimate that municipal water supply companies have spent a total of € 205 to 214 million on investments for drinking water treatment, as well as € 21,6 to 39 million in annual running costs. Of this, by far the largest part was spent on treating nitrate contamination, accounting for 72% of investments and 76 to 84% of annual running costs. In addition, € 11.7 million of compensation was paid to farmers who had reduced their nitrate appli-

[33] A point to note is that groundwater treatment for use as drinking water is different from groundwater remediation, as described in chapter 4.2. Although both employ purification technologies, which may have similar specifications, the goal of groundwater remediation is to improve the quality of the water *in the aquifer*, whereas the treatment discussed here only applies to that part of groundwater which is fed into the public water supply network. In this sense, the distinction between the two approaches is analytical rather than technological.

cations. The remainder of the cost largely stems from elimination of pesticide residuals, as well as a small amount (<2% of the total) for the treatment of chlorinated hydrocarbons.

Lacroix and Balduchi (1995) have estimated the cost of nitrate treatment for drinking water supply. They consider 25 denitrification plants that have been installed in France since 1981, exclusively in areas with high nitrate pollution from agriculture. Except for four cases, all plants were used to treat contaminated groundwater only. They find that for all plant locations, different options for ensuring safe drinking water supply had also been considered. The option of transporting drinking water in other regions had been rejected partly because of prohibitive costs or technological obstacles, but also because communities wanted to retain an independent drinking water supply. Pollution prevention strategies were disregarded because of lack of information and skilled personnel, and their lack of power to change agricultural practices. Therefore denitrification treatment was chosen as the easiest available option that would deliver immediate results. At the same time, it is clear that the decision was not based on cost-efficiency considerations alone.

The authors compare two different treatments for nitrate pollution: the biological denitrification (*dénitrification*) and the physical-chemical denitration (*dénitratation*), which is much more common in France. Both technologies are capable of reducing nitrate contamination almost entirely, however, they are commonly used only to reduce it to the recommended level of 25 mg/l. Concerning the investment costs, they find that, on average, biological denitrification requires much higher total initial investments (870,000 € / 340,000 €), but achieves a lower cost per unit of treatment capacity. This can be explained by the fact that biological treatment is more suitable for large-scale plants, which therefore achieve lower per-unit costs through economies of scale. As for the running costs of the treatment plants, the authors concur that such information is hardly ever publicised, since the plants are run by private firms in a very competitive markets. Also, the running costs are highly site-specific, as they depend on the pollution level and other local characteristics of the water to be treated. However, on average, there is not much divergence between running costs for the different procedures; both are around 12 ¢ per m^3. For the total costs, which combines investment, depreciation and running costs, the biological treatment is slightly cheaper (24 ¢ as opposed to 28 ¢ per m^3 for physical-chemical treatment, which equals 19,20 € / 22 € per inhabitant per year). They therefore conclude that both kinds of treatment have only marginal economic consequences - which, paradoxically, are not borne by the polluters, but by the victims. Indeed, all but two of the affected water suppliers admit that the installation cost of treatment plants have been passed on to consumers through higher prices. As the authors argue, the inequity of this situation is further increased by the fact that investment subsidies are granted for the plants, so that both water consumers and taxpayers contribute to them.

Altogether, Lacroix and Balduchi conclude that treatment offers only a partial solution, since it only applies to drinking water and only to nitrate contamination. In addition, it offers only short-term relief: if rising nitrate concentrations are not targeted, the costs of treatment will also rise and in some cases become prohibitive; therefore pollution prevention is indispensable in the long term. Concerning the cost of pollution prevention, as compared to the cost of treatment, Lacroix and Balduchi cite some evidence (De Haen 1990, Bel et al. 1995) that prevention is the cheaper option in almost all cases - only under very unfavourable conditions will the two costs be roughly equal for the same reduction target.

Box 6.5: Cheap Drinking Water from Forest and Floodplains

Küchli and Meylan (2002) present some figures on the relevance that forests in groundwater catchment areas have for the protection of groundwater. They argue that forests play a significant ecological and economic role in protecting groundwater resources because of their large potential for filtering out nitrates from precipitation and runoff. They report that, in the case of Switzerland, between 41 and 51% of the groundwater can be fed into the public water supply network without any prior treatment; for an additional 40%, a simple one-stage-treatment is sufficient. This is a marked difference compared to the 17% of drinking water supply that use water from lakes; this water has to undergo extensive chemical, biological and physical treatment. The authors estimate that, if the annual costs of water treatment (at € 89 million for 655 million m^3 of treated water, or 14 ¢€ per m^3) were applied to all groundwater used for drinking water supply, this would raise total costs by € 54 million annually. Since a large proportion of the groundwater which can be used without any treatment stems from sources with forested catchment areas, this sum can be seen as a very rough estimate of the value of forests in drinking water provision. Consequently, it shows that the cost of groundwater pollution would be even higher, if some of it was not retained by natural ecosystems (cf. chapter 6.4)

Kosz (1996), in an analysis of the economic value of the Danube floodplains east of Vienna, also considered their contribution to groundwater protection. He therefore considered the avoided costs of groundwater protection measures that would be necessary if the Danube was used commercially. Estimates of the investment, operation and maintenance costs necessary for securing groundwater quality ranged from € 44 to € 105 million if the river was used for hydropower generation. Kosz concludes from this that protecting the Danube floodplains in a national park can be recommended as the "most efficient way to protect groundwater quality and quantity" (cf. also box 6.6 for a discussion of the study, and chapter 6.4 on the estimation of ecosystem benefits).

6.2.3 Cost of Illness Approach

A third procedure used to assess the damage avoided by groundwater protection is the cost of illness approach. Like the Averting Behaviour studies, it only considers the health impact of contaminated groundwater and blends out other effects. The measurement of the cost-of-illness is relatively straightforward; it encompasses the cost of treating the illness and the income foregone through sick leave. However, there are a number of uncertainties associated with this approach:

- Consumers of groundwater will reduce their health risk exposure with averting actions. Obviously, if consumers perceive a serious threat from contaminated groundwater, they will discontinue using it and turn to other sources of drinking water; hence the averting behaviour and cost-of-illness approaches are to some extent complementary (Whitehead and Van Houtven 1997).
- The official quality standards for groundwater pollutants are set so that any danger of adverse health effects can be positively ruled out - hence the cost of illness approach will only be applicable if limit values are violated.
- The healthcare systems in most industrialised countries are largely state-funded, which means that there is no readily available market price for many treatments. Also, since up to two thirds of the total cost of treating illnesses are borne by third parties such as health insurances and employers, the social cost of illness exceed the private costs by far; however the former are much harder to quantify.
- As for the cost of foregone income, measurement is problematic for people not working in the official labour market, such as people who work at home. In order to calculate the full costs of illness, different estimation procedures have to be applied to value these non-markets goods.

In a survey paper on valuation approaches, Whitehead and Van Houtven list only one study that has investigated the cost of illness from contaminated groundwater (Harrington et al. 1989). This study was conducted in Pennsylvania following an outbreak of giardiasis and was restricted to those households that were actually affected. They estimated costs between US$ 1296 and 1895, of which US$ 384 were out-of-pocket medical expenses paid by the victims. There is, to our knowledge, no study of the cost of illness from persistent groundwater contamination, e.g. with nitrates or pesticides.

6.2.4 Conclusions on Avoided Damage Cost Approaches

In comparison to the second main type of measurement methods, the analysis of willingness to pay, many economists regard estimates based on avoided damage costs as more reliable. This is because these estimates rely on decisions that have actually been made, in contrast to willingness to pay, which is based on hypothetical surveys (cf. following section).

On the other hand, if the method is to produce coherent results, it must be ensured that the avoided costs are indeed primarily caused by concerns about the quality of groundwater (in its use as drinking water). While this is evident for avoided treatment costs in the public water supply, it is far less clear in the case of averting behaviour and the cost-of-illness approach. In these cases, it is also possible that factors other than concern about water quality play an important role, e.g. taste for bottled water, or, in the case of the costs of illness, other health risks. For these two methods, reliable estimates can therefore only be made for isolated and temporary episodes with a strong health impact. By contrast, the costs of ongoing and widespread low-level health risks, e.g. through high nitrate levels, cannot be assessed with these methods.

One limitation of all avoided damage cost approaches is that they consider only selected use-values of groundwater. The cost of illness and the averting behaviour approaches even focus exclusively on the provision of drinking water. Moreover, the value of groundwater protection is estimated only on the basis of the groundwater that is actually abstracted – and, by extending the results over time, also the water that will be abstracted in the future. However, the fact that a pollution reduction would also benefit the much larger part of groundwater that is not used, is not considered.

6.3 Benefits Estimated as Willingness to Pay

As explained in chapter 6.1, one way of valuing unpolluted groundwater is to let individuals or households state their willingness to pay for proposed measure that would improve groundwater quality. Such estimates are typically made through *contingent valuation* survey, whereby consumers in an area are asked to state their willingness to pay through mail survey or interviews. In order to receive more realistic answers, it is typically suggested that the interviewee would be obliged to pay the amount he or she offers to give.

A strength of this approach is that it directly addresses public concerns about groundwater pollution. These concerns can but need not be related to health risks; depending on the formulation of the question and the proposed measure, it is also possible to estimate the non-use values that consumers place on clean groundwater. Contingent valuation surveys have been very popular among economists, since they effectively simulate a market where "clean groundwater" can be bought as a hypothetical good. Another benefit is that a high willingness to pay can be interpreted as an indicator that the existing regulation is inefficient from a social perspective. Because of these advantages, contingent valuation surveys are so far the most widely used instrument for assessing the value of groundwater.

At the same time, some caveats apply. In practice, the idea of having to pay for improvements in groundwater is often rejected outright by the surveyed households - indeed, it is normal to perceive unpolluted groundwater as a natural right; others would feel that it is not their duty to pay for pollution caused by someone else. As a result, respondents state that they would not be willing to pay anything at all - although this need not imply that they do not value cleaner groundwater. There is some academic debate on whether such protest bids should be included in calculating the total economic value of groundwater. In most of the case studies below, they were excluded.

A second difficulty concerns the issue of benefits transfer: a survey necessarily includes a limited number of households only. It is debatable to what extent the findings from the survey can be generalised to all households in the country. In particular, the findings will differ if the survey has been conducted in an area where groundwater pollution is a problem and where, one can consequently assume that the knowledge of pollution and its effects is above the average. From a European perspective, the question of benefits transfer is especially relevant since few contingent valuation studies exist from European regions (with the notable exceptions of Press and Söderqvist (1998) (Milan), Stenger and Willinger (1998) (Alsace) and Tervonen (1994) (Oulu)). The issue of benefits transfer will be discussed in more detail in chapter 6.3.3.

6.3.1 Case Studies from the US

In a survey conducted in Dougherty County in Southwest Georgia, Sun, Bergstrom and Dorfman (1992) found a substantial willingness to pay for cleaner groundwater. This result was robust across all income ranges. Households were willing to give up roughly 2% of their annual income in exchange for reduced pollution from agricultural chemicals (fertilisers and pesticides); in 1992 US$, this equals an annual mean willingness to pay of US$ 641. As expected, the willingness to pay varied with certain parameters: it was higher for households with a higher exposure level, or with a higher likelihood of being affected; it was also higher for those people who stated that they were very concerned about their health; and it was higher for younger people. Sun et al. conclude that the high value placed on cleaner ground water indicates a need for political action. Without doubt, a programme to reduce groundwater pollution would improve social welfare - even if households in the area had to pay for it, they would perceive it as an improvement compared to the current situation. On the other hand, the authors also underline that their findings depend on the local circumstances of the area studied, and rather than transferring them to other areas, one would have to look for tailor-made solutions for other places with other circumstances.

Crutchfield, Cooper and Hellerstein (1997) conducted a survey on households' willingness to pay for reduced nitrate exposure in four areas in the US (White River / Indiana, Central Nebraska, Lower Susquehanna, Mid-Columbia basin / Washington). In their survey, they asked households to state their willingness to pay for a (hypothetical) water filtering device, which would reduce the risk of nitrate exposure to legal safety standard levels. For households in those areas where the current nitrate exposure exceeded the minimum safety standards of the US Environmental Protection Agency, the result was that households would be willing to pay 45 to 60 US$ per month to have exposure levels reduced to the safety standards. If this result is transferred to all households in the area that are potentially at risk, this implies an estimated benefit of 350 million US$ annually.

Poe and Bishop (1999, 2001) analyse the willingness to pay for reduced nitrate contamination in Portage County, Wisconsin. Their approach differs from the other empirical studies discussed here, in that they also consider the effect that information levels have on the valuation of groundwater resources. Their reasoning is as follows: the valuation of unpolluted groundwater is an unfamiliar exercise for almost any household; therefore, they provided households both with general background information on the different effects of nitrate pollution in drinking water, and with specific information on the actual exposure levels that the individual household faces. Unfortunately, this set-up implies that the individual willingness to pay results cannot simply be aggregated; Poe and Bishop therefore do not report a mean value.

One of their main findings was that the behaviour of respondents, and thus their willingness to pay, is subject to the level of exposure. For higher exposure levels, households would turn to averting behaviour, e.g. by using bottled water instead of tap water for drinking and cooking. If households chose complete aversion, it would mean that their health risks became independent of pollutant concentrations in groundwater. Hence, the possibility of averting action reduces the willingness to pay for better groundwater protection, especially at higher contamination levels. A second reason why willingness to pay may fall at higher contamination levels is what the authors refer to as the "indivisibility" of pollution: once a high level of pollution is reached (the threshold level mentioned for nitrates is around 15 mg/l), the water is considered "unsafe" anyway, so that small reductions of pollution are not perceived as an improvement – the improved water would still be unsafe. In accordance with this, Poe and Bishop found evidence of non-convexities in the benefit functions of respondents - in other words, they estimate that the willingness to pay first increases as exposure to groundwater pollution grows, but then declines from a certain threshold level onward. Thus, according to Poe and Bishop and contrary to conventional economic assumptions, the benefits of reduced pollution may actually diminish at higher pollution levels.

When singling out the effect of information on the willingness to pay, Poe and Bishop find that improved provision of *general* information leads to an increased willingness to pay, but also to an increased variance of replies. This can be seen as evidence that information is processed by different individuals in very different ways: households with small children, for example, will react much more strongly to information on the nitrate-related health risks for babies. However, when it comes to the provision of *specific* information – i.e. information on the actual level of contamination for each household – the situation changes: this information lowers the mean willingness to pay partly and thereby partly offsets the effect of better general information. Poe and Bishop conclude from this that providing only general information may inflate willingness to pay and lead water users to demand protection levels which are not in their best interest.

Epp and Delavan (2001) estimate the willingness to pay for reduced nitrate contamination from different sources in two counties in Southeastern Pennsylvania. One peculiarity of their survey was that they did not present the success of the proposed measures as guaranteed. Rather, they included peoples' expectations about the effectiveness of the programme as a variable in their survey. Depending on the type of questionnaire used,[34] the mean monthly willingness to pay for the proposed programme ranged from 54 US$ to 74 US$ per household. These results apply when protest bids, filled in by respondents who objected to the survey altogether, were eliminated. When they were not eliminated, this lowered the mean willingness to pay considerably – in fact it was not even possible to assert that willingness to pay was significantly different from zero.

In their survey, Epp and Delavan identified a range of factors that influence willingness to pay. Factors with a positive impact are household income, the perceived effectiveness of the programme, and previous action on the household level to avert pollution (e.g. households that had previously used bottled water or water filters were more likely to make a higher bid; this variable can be seen as a proxy for concern about clean groundwater). Somewhat paradoxically, the willingness to pay of households which had their own drinking water well was lower than that of households with municipal water supply, even though they were all supplied from the same aquifer. Epp and Delavan's speculative explanation is that private well-owners might hold the erroneous impression that groundwater protection would fall under their own responsibility. And, surprisingly, the presence of children in the household did not have a statistically significant effect on willingness to pay; neither did gender or age of the head of household.

In addition to the case studies presented above, the following table is based on a summary paper by Whitehead and Van Houtven (1997) and a survey compiled by Söderqvist and Press (1994); it gives an overview of contingent valuation studies prior to 1996:

Table 3

Contingent Valuation Studies of Safe Drinking Water

Study	Location	Commodity Valued	Sample Size	WTP (a) (in US$)
Edwards (1988)	Cape Cod, Massachusetts	An aquifer management plan to reduce the probability of nitrate contamination	585	$ 155
Schultz and Lindsay (1990)	Dover, New Hampshire	Protections plans to protect community groundwater supplies	346	$ 15

[34] The questionnaires used contained both open-ended questions and dichotomous choice question. With open-ended question, the interviewees are asked to state how much they would be willing to pay for the proposed measures, whereas dichotomous choice questions are yes-or-no-questions, asking whether the interviewee would vote for a proposal if it implied an additional annual cost of x or y €. The bid levels used in the questionnaire are typically derived from previous pretests.

Sun, Bergstrom, and Dorfman (1992)	Doughtery County, Georgia	Protecting "safe" groundwater from potential future contamination	603	$ 67
McClelland, et al. (1992)	National (US)	Complete groundwater cleanup from a 40% contamination	1983	$ 12
Caudill and Hoehn (1992)	Michigan	Action to prevent contamination; maintenance of well water quality	1213	$ 65
Poe and Bishop (1992)	Portage County, Wisconsin	Groundwater protection program to prevent nitrate contamination	244	$ 24 (b)
Jordan and El-nagheeb 1993)	Georgia	Preventing groundwater pollution that would make sure water is safe for drinking	180	$ 14, $ 16 (e)
Powell, Allee, and McClintock (1994)	Massachusetts, New York, and Pennsylvania	Establish water supply protection districts that would ensure safe drinking water	1021	$ 6
Laughland, et al. (1996)	Milesburg, Pennsylvania	Connection to an alternative source so that drinking water meets standards	226	$ 23
Clemons, Collins, and Green (1995)	Martinsburg, West Virginia	Wellhead protection program to eliminate risk of contamination	576	$ 1,8, $ 1,2 (d)
Krug (1995)	Western Massachusetts	Aquifer Protection District and purchase of a private water filter	397	$6, $7 (e)
Power et al. (1991)	12 communities, north-eastern USA	Increase water supply protection	1006	$ 61,55
Kwak and Russel (1994)	Seoul, South Korea	Government plan to reduce probability of major contamination incidents to zero or near zero	298	$ 40

(a) Monthly household WTP (1996 dollars) is adjusted using 4.3 weeks/month, 30 days/month.
(b) With information sample.
(c) WTP is for private wells, municipal sources.
(d) WTP for nitrate, VOC contamination.
(e) WTP for protection district, private water filter.

Source: Whitehead and Van Houtven (1997), Söderqvist and Press (1994)

6.3.2 European Case Studies

Press and Söderqvist (1998) offer an estimate of willingness to pay for groundwater protection in an aquifer underlying the Northern Italian city of Milan. The aquifer had previously experienced atrazine contamination from agricultural emissions, albeit not at critical levels – indeed it is unclear whether the observed concentrations would imply a significant risk to human health. The analysis was concerned with the willingness to pay for the introduction of a comprehensive groundwater management plan. Interviewees were informed that a failure to implement such a plan would imply a further degradation of water quality in the aquifer, and would therefore mean that European drinking water standards would no longer be met for water supplied from the aquifer. The analysis combined dichotomous choice and open-ended bids (cf. footnote 33).

The analysis is particularly interesting for two reasons:

- At 0.13 €/m^3, the price of drinking water in Milan is one of the lowest in Europe, at only one sixth of the European average;
- Nonetheless, Northern Italy is among the regions with the highest bottled water consumption per capita; more than 70% of the respondents stated that they used bottled water as their exclusive source of drinking water.

Based on different calculation measures, the mean annual willingness to pay derived from the dichotomous choice analysis was reported between 425 and 559 € per household, based on 144 observations. The open-ended questions resulted in a mean WTP ranging from 215 to 231 €.[35] These results are quite high: they imply that the mean willingness to pay amounts to 1,2 % of household income, or 166% of the annual expenditures on bottled drinking water. This indicates a willingness to pay that is higher than in comparable studies. The explanations offered by the authors for this divergence include an exaggerated perception of health risks, sparked by media reports; but also, at least partly, concern for the non-use values of groundwater protection. This is supported by the fact that 96% of respondents stated that they view groundwater protection as important in its own right, apart from concerns with the safety of drinking water supplies.

Stenger and Willinger (1998) present a contingent valuation study of the willingness to pay for preservation of the Alsatian aquifer, one of the largest aquifers in Europe, which is located in the Northeast of France. The water of the Alsatian aquifer is of high quality – in most places water is drinkable without treatment – and it is accessible at low cost. The authors estimate the total economic services of the aquifer at € 50.3 million annually, by providing 80% of the drinking water in the region, and 54% of the industrial water supply. Remarkably, industrial groundwater use is three times higher than that of private households, which means that the industrial water demand is twice the French average. However, in recent years the aquifer has increasingly been threatened by pollution from mining, industry, agriculture and transport (cf. also the case study on Alsatian potash mining fields, BRGM 2003). Next to rising nitrate concentrations from agriculture, salts from Potash mining have been identified as major polluters.

Stenger and Willinger particularly set out to test three hypotheses from earlier, American contingent valuation surveys, namely that (1) exposure to previous incidences of contamination would increase WTP, (2) WTP was positively correlated with the reliability of the preservation programme, and (3) additional information about the water bill would raise WTP. The hypotheses were tested with a survey of 800 households in ten municipalities, three of which had been exposed to groundwater contamination. The mailed questionnaires tested WTP both as dichotomous choice and as open-ended question, offering a programme that would bring down the risk of contamination by almost 100% to some households, or by 75% to others. Notably, the proposed programmes were not designed to protect drinking water from one particular contaminant (as in most cases above), but to protect the aquifer from pollution altogether. The authors found a WTP of € 94 per household per year. Of the sample addressed with dichotomous-choice proposals, just below 70% agreed to the proposed programmes, and even at the highest bid of € 152 per year, 57% voted for the programme. Concerning their hypotheses, Stenger and Willinger found that indeed WTP was higher in communities that had been affected by pollution in the past. Surprisingly, the reliability of the programme (at 75% or at 100%) did not have a significant impact on willingness to pay, which led the authors to reject their second hypothesis. Finally, they also found no significant evidence that respondents would state a higher WTP after they had been presented with information on their annual water bill.

Tervonen et al. (1994), in one of the first European contingent valuation studies in the water sector, investigated the willingness to pay for unpolluted *drinking water* in the Finnish city of Oulu. Their analysis is only of indirect relevance for the value of groundwater: in contrast to the other analyses presented in this section, they did not consider the demand for *groundwater protection measures* as a means of safeguarding water supply. Instead, they considered willingness to pay for alternatives to the current drinking water supply, which is purified river water from the river Oulu. The interviewees were presented with two different options of improving drinking water quality: either to change to more sophisticated treatment technology, or the much more costly option of importing groundwater from a nearby aquifer. Presented with bids for improving supply ranging from 2 to 80 €, consumers reported an av-

---

35 Please note that the original amounts stated in 1998 Italian Lira have been converted to Euro, but not discounted. The considerable difference between the results from the two procedures points to a sizeable *anchoring effect*, whereby the stated willingness to pay depends on the proposed bids themselves. Comparing the different estimation procedures, the authors argue that an average WTP of 449 € could be seen as the most realistic result.

erage willingness to pay of about 51 € for enhanced purification of river water, compared to 54 € for groundwater.[36] Both of these average bids far exceeded the actual cost that would be expected from these measures (8 € and 38 € respectively). The authors interpret the remarkable closeness of the two results as an indication that consumers are mainly concerned with the quality of drinking water supply, while the actual source of the water is largely irrelevant to the valuation. In a simple cost-benefit calculation, they subtract the expected annual cost per capita from the average stated willingness to pay. The conclusion is that, obviously, both options are feasible and would lead to a significant net increase in welfare; however the expected increase is almost three times as high for the enhanced purification solution because of its lower cost.

Box 6.6: The Danube Floodplains and the Value of Groundwater

Kosz (1996) presents a contingent valuation study of the proposed *Donauauen Nationalpark* (Danube floodplains national park), to be established in the Danube valley east of Vienna, Austria. Kosz' analysis differs somewhat from the other papers treated here, since it is not primarily concerned with groundwater. Instead, potential users were asked how much they would be willing to pay for the establishment of a national park, instead of designating the area for commercial uses (hydrolectric power generation or shipping). The information provided to the respondents beforehand also mentioned, as one of four aspects, the groundwater-enhancing functions of floodplains. However, it is not clear to what extent respondents have considered this in their responses, or what relative weight they might have placed on groundwater enhancement and protection. This means that the results from the contingent valuation are at best a very indirect estimate of the value of groundwater.

6.3.3 Conclusions on Analyses of Willingness to Pay

The range of contingent valuation studies listed above indeed confirms that there is a significant willingness to pay for improvements in groundwater quality; hence, in the municipalities affected, stricter protection measures would enhance social welfare. Yet policy recommendations beyond the studied area are not easily derived from this: the willingness to pay has been calculated on the basis of regional surveys, taking into account highly localised demographic and geological parameters. The findings are therefore applicable only to the region where the survey was conducted - even more so if the region was chosen *because* it was affected by exposure to groundwater contamination above the national average.

For the valuation of groundwater resources, it would be very helpful if a mechanism could be devised, by which findings from a set of studies could be transferred to other locations. Such an approach would first identify the relevant factors influencing willingness to pay, such as income, family size, age structure, educational level etc. It would then calculate how these factors influence willingness to pay, based on a number of different surveys. In the ideal case, the willingness to pay for the target region could then be calculated by simply putting in the relevant values - which are usually readily available from statistical yearbooks. However, this transfer of findings from one study to other regions is fraught with methodological difficulties - to take an extreme case, if willingness to pay depends on the educational level of the head of household, should the groundwater beneath university cities be valued higher? In addition, the empirical base is still considered too small and therefore too unreliable to allow for such a benefit transfer (Boyle et al. 2001). This is even more evident for the non-use values of groundwater: whereas the value of groundwater for drinking water provision has been covered fairly extensively, much less is known about the willingness to pay for clean groundwater if there is no (immediate) intention of using it.

One particular strength of willingness to pay analyses is that they allow a valuation of groundwater that is fairly close to consumers' preferences. For example, Söderqvist (1998)

[36] Unfortunately, the authors do not indicate whether there is a systematic preference for drinking water supplied from groundwater. However, since the willingness to pay for groundwater is only 6% higher than for purified river water, this difference is probably not statistically significant.

quotes a study by Vighi and Zanin, who found that the health risks from atrazine (pesticide) levels in groundwater were virtually zero. Nonetheless, Södeqvist found a substantial willingness to pay for measures that would reduce atrazine concentrations even further. Although this behaviour may seem irrational from a scientific perspective, it also shows that the relevant policy target need not be determined on scientific grounds alone. If consumers perceive risks from contaminated groundwater differently from scientific experts, and are willing to pay for reduced exposure levels, this still indicates that less pollution would be more efficient from a social point of view. Whether consumers are indeed irrational, or whether they have reason to doubt the scientific certainty, is irrelevant for the finding that potential benefits can be achieved from better groundwater protection levels.

6.4 Ecosystem Benefits

As stated in the introductory section 6.1.4 above, the concept of ecosystem benefits relates to the effects that groundwater has on groundwater-dependent ecosystems. From this perspective, the benefits of groundwater protection consist of the *avoided damage* that polluted groundwater would otherwise cause to groundwater-dependent ecosystems, such as wetlands, lakes and rivers. This means that, in order to assess the ecosystem benefits, a monetary estimate is required of how the economically relevant services of the ecosystem are affected if the quality of the discharged groundwater deteriorates. In other words, the calculation of the ecosystem benefits requires two pieces of information:

- the value of the ecosystem itself (expressed as the sum of all ecosystem services), and
- the impact of groundwater contamination on these ecosystem services.

Obviously, both questions are in themselves difficult tasks; on the one hand, the valuation of a groundwater-dependent wetland could be done using the same procedures that were described above for the case of groundwater. The interaction between groundwater and surface ecosystems, on the other hand, has attracted the attention of environmental economists only in recent years. Consequently, very little theoretical work has been published on the ecosystem benefits of groundwater protection (see e.g. Abdalla 1994), and even less on empirical evidence.

Box 6.7: Putting a Price on Wetlands

Turner et al. (2000) have considered the question of valuing the services provided by a wetland from an ecological-economic perspective. They argue that the economic valuation of wetland services could be described as the step from wetland *functioning* to wetland *uses*. Wetland uses are restricted to those wetland functions to which society attaches some sort of value.[37] Turner et al. identify eleven such uses, ranging from amenity or fuelwood provision to nitrate retention and biodiversity maintenance. However, when turning to the valuation of these wetland uses, they point out that the available knowledge is still far from sufficient to come to definite conclusions. “In particular, to predict in detail a policy’s impact on such wetland functioning as, for example, nutrient and sediment retention, gas exchange, and pollution absorption, for any given segment of landscape, is in many cases likely to push present ecological knowledge beyond its bounds.“ (Turner et al. (2000), p. 14).

Gren and Söderqvist (1994) offer a survey of 30 studies from North America, Europe and Asia that have estimated the value of wetlands in monetary terms. Many of these studies have done this by way of a contingent valuation analysis, others by estimating what the services provided by wetlands would cost if they were provided artificially. Unfortunately, none of the studies contains explicit references to the effects of groundwater on the value of a wetland.

[37] Unfortunately, there is a slight difference in terminology here: if the criterion for a wetland use is whether any sort of value is attached, then non-use values would also qualify as wetland uses.

The different studies surveyed by Gren and Söderqvist mainly considered the following services:

- flood control,
- water supply,
- recreational and amenity value,
- nutrient removal and retention of toxic substances, and
- maintenance of biodiversity and wildlife

The reported total annual value differed strongly between the various studies, depending on the methods used and on the ecosystem services included in the analysis. Overall, reported estimates range from US$ 1.2 to 39,777 per hectare, where results based on willingness to pay for use- and non-use values were much lower than analyses that considered specific environmental services, and valued these at the cost of alternative supply options. The 11 European studies contained in the sample arrived at somewhat lower values, ranging from US$ 34 to 1300 per hectare and year. A detailed survey of the results from the European case studies is given in the table below.

Table 4
Estimates of the Value of Wetlands

Author	Region	Environmental services valued	Results (measured in 1993 US$ per year)
Folke 1991	Gotland, Sweden	nutrient sink, fish, water supply	240 / ha
Hanley and Craig 1991	Scotland	use, non-use	34 / ha Total sample: 1.7 / person Users: 2.6 / person Non-users: 1.3 / person
Tomasin 1991	Po delta, Italy	fish, hunting, recreation	1,500 / ha
Kosz et al. 1991	Vienna, Austria	forest production, grassland, fish, recreation	> 522 / ha
Bateman et al. 1993	East Anglia, UK	recreation	108 - 266 / person
Ecotec 1993	Aquatic ecosystems, UK	use, non-use	Total: 42 / household Users: 52 / household Non-users: 28 / household
Gren 1993	Stockholm, Sweden	nitrogen sink	430 / ha
Gravener 1994	Oxelösund, Sweden	use (except nitrogen sink), non-use	570-1,150 / ha 1.3-585 / person
Gren 1994a	Gotland, Sweden	nitrogen sink, fish, water supply	239-585 / ha
Gren 1994b	Danube (floodplains)	nitrogen sink, forest production, fish, hunting, grassland, recreation	458 / ha
Ungerman 1994	Nové Mlyny, Czech Republic	forest and agriculture production, hunting	290 / ha

Source: Gren and Söderqvist (1994). Note that the size of the researched wetlands ranges from 22 ha in the study by Gravener (1994) to 202,000 ha in the study by Hanley and Craig (1991).

As the Box above shows, some research has been done on the valuation of ecosystems in general, and on wetlands in particular. However, there appears to be no empirical study so far which examines the contribution from groundwater to the monetary value of a wetland, and shows how this value is affected by groundwater contamination.[38] Therefore we cannot offer economic estimates of the ecosystem benefits of groundwater protection.

[38] The absence of empirical evidence on this issue was also confirmed by Tore Söderqvist of the Beijer Institute, Stockholm, and with Robert Cunningham of the British Wildlife Trusts' Water Policy Team (personal communication, January 2003).

A particular caveat with respect to the valuation of ecosystems in general, and to the value of wetlands in particular, has been put forward recently by Balmford et al. (2002). In a survey article, they review the evidence on the economic value of different ecosystems, particularly with a view to ecosystem and local cultural benefits. Unfortunately, they do not refer to groundwater in their article. However, for all the cases they investigate, they find that the economic gains of human conversion of these ecosystems (in order to use them for agricultural or other economic purposes) are actually negative. For the case of a Canadian wetland, the total economic value actually decreased by more than 40% as a consequence of conversion (from US$ 8800 to US$ 3700 / ha / y). The reason for this surprising finding is that the loss of the non-marketed services provided by the ecosystems is not outweighed by the marginal benefits of conversion. The finding holds despite the fact that some particularly valuable ecosystem services, such as nutrient cycling or the provision of cultural values, were not considered due to a lack of data.

A preliminary assessment of the overall relevance of ecosystem benefits in the context of groundwater protection can be tentative at best. In many aquifers, groundwater abstraction for human use affects only a very limited proportion of the total groundwater body. By contrast, the quantities discharged to rivers, lakes and wetlands are frequently significantly larger than human abstractions. This consideration by itself does not imply any economic ranking; however, it should be considered as a parameter guiding the site-specific analysis of groundwater values: if the analysis is concerned with an aquifer that is largely isolated from surface water bodies, and where there are no groundwater-dependent ecosystems of a high economic value, the ecosystem benefits may be negligible. By contrast, if there is much exchange between groundwater and surface water, and if there are groundwater-dependent ecosystems of a high economic and ecological value in the study area, failure to include them implies a significant underestimation of the total benefits of groundwater protection.

In the absence of further empirical economic studies of these issues, it is not possible to place a general value the ecosystem benefits, or to assess their relative importance in comparison to the direct benefits of groundwater protection (use- and non-use values). However, taking the precautionary principle into account, two points must be strongly emphasised:

- The fact that no valuation of ecosystem benefits is available does not mean that their value is zero. This point is especially relevant when it comes to correcting the "measurement bias" inherent in the valuation of natural resources: the fact that the costs of preservation and protection are manifest and quantified in monetary terms, whereas the benefits are often less tangible, less visible and less concrete (cf. chapter 6.1.1). This bias applies to ecosystem benefits even more than it does to use- and non-use values.
- The valuation of ecosystem benefits has to take into account that currently unused resources may be used in the future, that groundwater which is currently in abundant supply may become scarce in many regions within decades, and that damages to groundwater and dependent ecosystems may be irreversible. Also, for most ecosystem benefits, substitution with other sources than groundwater will not be a feasible alternative, which sets the ecosystem benefits apart from direct uses of groundwater.

Box 6.8: The Relevance of Ecosystems for Groundwater Protection

An interesting aspect of the relationship between groundwater and ecosystems is the fact that many ecosystems play an important role for the protection and recharge of groundwater bodies. Technically, these functions must be distinguished from ecosystem benefits: they concern the role of ecosystems for groundwater quality, whereas ecosystem benefits describe the effect of groundwater quality on dependent ecosystems. Technically speaking, ecosystems therefore are regarded as part of groundwater protection and remediation, thereby reducing the cost of treatment for polluted groundwater.

Different examples of this were discussed in previous chapters:

- Box 6.5 discussed a study from Switzerland that estimated the value of forests for the retention of nitrates and toxic substances, which would otherwise be leached to the groundwater. For Switzerland, this function of forests alone was estimated at an annual value of € 54 million per year.

- Likewise, the value of the Danube floodplains for groundwater protection and recharge was calculated by Kosz (1996). His study estimated that alternative, man-made measures for groundwater protection would cost between € 44 and € 105 million in total – costs that could be saved if the floodplains were left intact. Indeed, conservation emerged as the most cost-efficient option.
- Chapter 3.3.1 presented some evidence of natural or constructed wetlands in Sweden, which were found to be a cost-efficient method for nitrate retention and groundwater recharge.

The evidence above shows that an economic assessment of groundwater protection should not only consider the ecosystems that are dependent on groundwater discharges, and how they are affected by groundwater pollution. In addition, it must also consider the value of ecosystems a as a cost-effective method of groundwater protection and recharge.

6.5 Conclusions on Groundwater Valuation

➢ ***Limited knowledge base***

From the evidence on groundwater valuation discussed above, it appears that research so far has focused primarily on a small section of all groundwater functions and services. Therefore the current empirical knowledge of groundwater values is still rather limited. Only a few issues have been researched extensively, such as the value of groundwater for drinking water use; in these areas a reasonable knowledge base exists. In other areas, especially in the field of ecosystem benefits, empirical evidence is tentative at best.

➢ ***Increasing uncertainty with increasing abstractness***

Reviewing the evidence available of different uses for groundwater and the various services it provides, it emerges that the evidence is less the more indirect and abstract the considered values and functions are. Thus, from the consumer's point of view, the provision of drinking water from groundwater is the most tangible and direct use of groundwater, and has consequently received most attention in economic studies. Far less evidence is available on the more abstract non-use values, and the most indirect concept, the ecosystem benefits of groundwater protection, is only covered theoretically. With the available evidence, it is not possible to say whether this prioritisation of direct, human uses is warranted by the fact that they are economically most relevant.

➢ ***Plenitude and diversity of estimation methods***

A factor that complicates the analysis is the plenitude of estimation methods for groundwater values, and their unclear relation. Abdalla (1994) finds that economists tend to specialise on one method only, and that some categories have received much more emphasis than others – for example, the majority of research has focused on the willingness to pay for reduced health risks from diffuse source pollution, whereas there is practically no evidence on ecosystem effects of groundwater pollution. Abdalla sees this compartmentalisation as an obstacle to the more effective and widespread application of economic knowledge to political decisionmaking; he therefore argues that future research efforts should be directed more to *integrated analyses* of different effects of pollution.

This view is supported by Whitehead (1997), who argues that different valuation methods should be seen as complementary rather than substitutive. He argues that a full estimate of the benefits of groundwater protection should integrate different perspectives, using different methods, and considering different circumstances. At present, however, no satisfying frameworks for such integrated estimation procedures have been put forward.

➢ ***Transferability of results***

Whether the broad experience with contingent valuation studies in the US already allows for a benefit transfer, i.e. the application of findings to other sites without further empirical surveys, is still subject of academic debate. Poe, Boyle and Bergstrom (2001) state that, even with the extensive experience available, they would be extremely cautious about using the results of their meta analysis to construct a benefit transfer function. For the EU, with only three such studies, much more research would be required before any conclusions are possible.

A central question from the European perspective is whether the results derived from US case studies can be transferred to Europe. As far as quantitative results are concerned, this would have to be approached with great caution. Household incomes, which have emerged as a central determinant of willingness to pay, are comparable for many European regions. However, for other factors, such as environmental awareness or consumer confidence in the safety standards for drinking water, without further research it is impossible to say whether they differ significantly between the two regions.

➢ ***<u>General conclusions on consumers' willingness to pay</u>***

While quantitative transfers should be viewed with caution, some general points can be inferred from the US experience:

- Willingness to pay is certainly more than statistical "white noise". There are solid relations for factors that influence willingness to pay, such as income, education, and environmental awareness. People do care about their groundwater, and are willing to pay for it. This effect is measurable, and the amounts stated are usually significant.
- There is substantial willingness to pay for non-use values as well: groundwater protection is a concern even if there is no intention to use it. If WTP analyses that consider use values only are used as a minimum estimate for Total Economic Value, there is a great probability that the actual WTP, including non-use values, would be substantially higher.
- Stated willingness to pay, as well as averting expenditures, are site- and situation-specific. Most studies conclude that the type of water supply, the cost of averting actions, as well as household and community characteristics have an impact on the stated WTP.

➢ ***<u>Unbalanced evidence on different types of pollution</u>***

One clear limitation of the available evidence on groundwater values is that practically all the empirical studies presented above either refer to *diffuse pollution* only (and here mainly to nitrate pollution), or to a general concept of unpolluted (drinking) water. In terms of benefit transfer, this brings up two further questions: whether it is possible to apply the valuation for nitrate-free groundwater to other kinds of pollution; and how to deal with multiple contamination problems. Short of any empirical research, this problem may not be resolved easily.

7 Combining Costs and Benefits

In order to assess the contribution economics can offer for groundwater protection, it is necessary to combine costs and benefits of different measures and/or policy options. This remains the most challenging aspect of integrating economic considerations in groundwater protection.

The possible theoretical methods for such a combination are presented first, while the next chapter gives an overview of practical cases in which such estimates were used to investigate the optimal use of particular instruments. In order to clarify the possible contribution of economics in the policy development in the field of groundwater protection, examples of Risk-Based Management of Groundwater Resources are then presented. The concluding chapter investigates the possibilities and limitations of economic assessments for setting target values of groundwater protection.

7.1 Methods for Combining Costs and Benefits

A number of procedures have been developed to assess the economic efficiency and the social desirability of different policy alternatives. From an economic perspective, their aim is to combine the information on the costs and benefits of different measures. The three main ways of doing this are through a cost-benefit analysis, through a cost-effectiveness analysis, or through a multi-criteria analysis. These methods will be explained in the following.

- The most extensive method for evaluating the desirability of different policy options is the ***cost-benefit-analysis*** (CBA). Its aim is to estimate the total cost of carrying out a proposed policy, as well as the estimated benefits that the policy will bring to different stakeholders. In order to be comparable, both have to be calculated in monetary terms. If this information exists for all possible alternatives, it is straightforward to chose the option that maximises net social benefits. Policy alternatives in these cases should be understood broadly: the process of setting an optimal pollution target through quality standards or tax levels optimally should also be the result of a cost-benefit analysis. Unfortunately, as was argued in the previous chapter, it is very difficult to arrive at reliable estimates for the benefits of groundwater protection policies - in opposition to the costs, where there is usually sufficient evidence.

- Taking account of the difficulties associated with putting a monetary value on benefits, the ***cost-effectiveness analysis*** (CEA) therefore abandons this requirement. Instead, it compares the costs of different policy options which all lead to the same, given target. In contrast to the CBA, the target itself is thus not determined through the analysis: it has to be set 'exogenously', i.e. through a political decision. Therefore, the CEA delivers a result that is optimal *given the politically set target*, while a CBA claims to define the socially optimal level.

- Finally, the ***multi-criteria analysis*** (MCA) consists of two steps: in the first step, a range of objectives in different dimensions are identified (such as environmental, economic and social objectives), and the trade-offs between these objectives are specified for different policy alternatives. In a second stage, the different options are compared by attaching weights to the different objectives. The determination of these weights is a crucial but very difficult element of the analysis: weights can be purely monetary (in which case there is little difference between a MCA and a CBA), but they can also be based on public participation.[39] A crucial difference between multi-criteria analyses and the other two options is that the MCA allows for different outcomes in terms of environmental effectiveness *and* costs.

[39] A more extravagant option is to attach a large number of random weights, and then to compare how this affects the ranking of the different alternatives, an example of this is discussed in Box 7.1 below.

A different alternative, which is closely related to multi-criteria analyses, is to combine different objectives using a pre-defined weighing scheme. In this way, results in different dimensions are standardised (e.g. by allocating points in each category), and then summed up according to a predefined procedure. One example of such a weighing scheme is the four-account-model, which will be explained below in box 7.2. Other examples are decision guidance systems on soil protection and remediation, which have some history in the Netherlands (cf. chapter 5.2). Different weighing schemes, typically based on a restricted multi-criteria-analysis, have been developed on the national level, by the City of Utrecht, and in the Provincie Zuid-Holland (see OVAM 2002 for an overview of the different approaches). In general, the advantage of such standardised approach is that they require less conceptual effort, and less primary data collection; at the same time, the fact that weights have to be allocated to the different dimensions necessarily introduces an element of arbitrariness, which will always make it vulnerable to criticism.

Since a full cost-benefit analysis is associated with a very extensive effort for collecting the required information, and consequently with very high costs, it makes sense to see the above measures as hierarchical steps in the evaluation process. This approach is put forward by the British Environment Agency, which recommends a stepwise approach to an economic analysis (Environment Agency 1999, 2000). Cost comparisons and preliminary assessments should be used to assess whether a full CBA is justified in the first place. Only if there is substantial doubt as to whether the costs of a measure are in line with the expected benefits, a full Cost-Benefit-Analysis should be considered.

At the same time, if there is universal consensus that the benefits of a proposed measure will outweigh the costs, or if the quality target itself is given beforehand, a cost-effectiveness-analysis or even a simple cost comparison of the different options are sufficient, since they both allow a ranking of different alternative solutions. In the context of groundwater, it appears that a cost-benefit-analysis is therefore an unsuitable instrument for assessing policy alternatives on a national scale, but that it should rather be used to assess whether temporary derogation from a general protection target is justifiable.

7.2 Examples of Combining Costs and Benefits of particular instruments

Because they are connected to an enormous effort for the gathering of data, full cost-benefit-analyses of groundwater protection are not very common. The following section presents some examples where cost and benefits have been combined in different ways. However, the analyses are limited in the sense that they do not estimate optimal target levels for groundwater pollution as such, nor do they consider the optimal mix of different instruments. Instead, they investigate the optimal use of particular instruments – taxes, catchment protection, or cooperative agreements.

Gramel and Urban (2001) offer a simplified analysis, which is rather a rough cost comparison than a cost-effectiveness-analysis (cf. chapter 4.1.3). Comparing the effectiveness of cooperative agreements for reducing agricultural nitrogen discharges with the costs of purifying groundwater for use as drinking water, they conclude that it is more efficient to conclude cooperative agreements for the protection of groundwater at an average cost of 0.29 €/m^3 of protected groundwater than to treat the abstracted groundwater afterwards (at a cost of 0.25 to 0.75 €/m^3). However, since they only consider treatment costs for abstracted groundwater (i.e. its use-value), the efficiency of groundwater protection depends on the volume of abstractions from an aquifer: for aquifers with a low abstraction volume it may not be a competitive option. At the same time, their reference cost is the treatment of groundwater to meet *drinking water* standards, which need not be a suitable policy target.

Fuchs (1994) offers an interesting calculation for the derivation of an optimal nitrogen tax level for an exemplary region in Germany. For the calculation of a hypothetical, optimal tax, the avoided treatment costs for municipal water supply companies are balanced with the foregone income for farmers who reduce their nitrogen application. The study considers four parameters:

- the amount of nitrogen fertiliser applied;
- the resulting nitrate leachate;
- the additional treatment costs that this entails for the water supply company; and
- the resulting optimal tax level.

These parameters are estimated for four different optima: the partial equilibrium from the farm perspective and from the water supplier's point of view; and two different joint equilibria, depending on whether the farm engages in cattle farming or produces market crops. The authors come to the following results:

Table 5

Calculation of optimal nitrogen tax levels

	Nitrogen fertiliser application	Nitrate leachate	Additional treatment cost	Optimal tax level
	kg N	Mg NO_3/l	€/ha*a	€/kg N
Partial optimum water supplier	0	38	76,7	
Partial optimum farm	320	319	1023	0
Joint equilibrium market crops	96	ca. 75	ca. 95	1,95
Joint equilibrium cattle farming	193	ca. 165	ca. 400	3,55

Source: Data Hofreither and Sinabell (1996) based on Fuchs (1994)

Note that the optimal solution from the water suppliers point of view would be no fertiliser at all, which would require a ban on nitrogen fertiliser – or a prohibitive tax. On the other hand, the optimal solution for the farm is to simply maximises crop yield, not taking nitrate pollution into account; this of course implies a tax rate of zero.

The study results, however interesting, are subject to two constraints:

- They only consider the water that is abstracted from a polluted aquifer for provision of drinking water; other uses of groundwater are not included.
- The study compares the costs that water supply companies face in order to bring drinking water to the legally defined quality standard – it does not consider whether these standards themselves are efficient.

Box 7.1: Multi-Criteria Analysis as a Way of Combining Costs and Benefits

Messner et al. (2001) present a case study which evaluates the use conflicts associated with groundwater protection zones in the Eastern German region of Torgau (cf. chapter 5.1). In the study region, almost two thirds of the surface area are designated as protected areas, which puts severe restrictions on economic uses. Messner et al. employ a multi-criteria-analysis to define the most relevant problems which this poses from an economic, social and environmental viewpoint. They then compare these effects for 4 alternative policy scenarios over a period of 30 years; the scenarios differ in terms of whether the protected areas are decreased or remain unchanged, and whether gravel production is increased or remains the same.

As a benchmark value, Messner et al. use the costs and benefits of the different scenarios in purely monetary terms. The benefits range from the avoided cost of averting behaviour by groundwater users, to the benefits for users of bathing lakes in disused gravel pits (refer to chapter 5.1 for a detailed description of the cost components). The surprising result is that the total discounted benefit of the scenarios with a foreseen *increase* in gravel production deliver a smaller benefit, ranging from -4,9 billion € to 0 (for the reference scenario); this is due to the fact that there is already an oversupply of gravel in the region. By contrast, the benefit in the scenarios without additional gravel production ranges from 9,5 to 9,9 billion €.

From this purely monetary analysis, the authors proceed to a multi-criteria analysis. On the economic side, in addition to the net benefit from the planned measures, they now also consider the gross value added. This measure that incorporates feedback effects and second-order economic benefits (i.e. increased spending on consumer goods in the region as a consequence of employment). Social effects were approximated by the number of people employed in the region. The environmental dimension was measured with the change in groundwater stocks, the nitrogen input as well as a qualitative estimate of the ecological side-effects of gravel production. The different results for the various parameters were assigned random weighed factors, and a comparison made between the likelihood that different scenarios would be ranked first (second, third etc.). Thus, although the question of weighing different benefits against each other was not answered conclusively, it emerged that a restriction of protected areas would hardly ever be an optimal strategy. By contrast, under most weight distributions, the scenarios with no change in protected areas would be the first- or second-best solution, the scenario with an unchanged number of pits scoring slightly higher on average. This can, therefore, be seen as a win-win situation where both the environment and the economy gain from increased protection, or, economically speaking, where the abatement costs of groundwater protection are negative.

Box 7.2: The 4-account model as an alternative framework

The four-account-model provides an alternative framework to be used in the efficiency analysis of groundwater protection. Although it offers interesting possibilities for groundwater valuation, it is not commonly used in environmental economics, and not well embedded in standard economic theory.

The four-account-model combines economic efficiency along with a range of other criteria that determine economic welfare, such as the social and regional distribution effects of the policy measures. Furthermore, it also admits environmental effects in non-monetary form.

The Four-account-model distinguishes between four different categories to be used:

I. Overall economic efficiency - comprises what is traditionally measured in cost-benefit analyses, i.e. the measurable economic effects in terms of income/output changes as a consequence of pollution, avoidance costs, and consumers' willingness to pay for higher groundwater protection levels.

II. Environmental quality - relates to a non-monetary assessment of environmental impacts, based on the relevant physical and ecological criteria

III. Regional development - allows for a detailed breakdown of the regional effects on income, demography and employment that a groundwater protection measure has. Thus, it puts in relief the different local impacts that may be averaged out in a larger-scale analysis.

IV. Social impact - contains the effects that a measure has on different social groups and income classes; as well as the effects on different economic sectors.

A crucial point is that these categories are measured on different scales, and comprise both economic and non-economic measures. This also means that values from different categories should not be added up into one common figure. The advantage of this approach is clearly that it allows a broader focus, taking factors into account that are of direct political relevance, but not usually included in economic analyses. The main problem with this approach is that it does not suggest how the results in the different categories should be weighed. Hence, the conclusions from this method remain subject to political debate.

Quadflieg (2002) summarises the results of applying this framework to the Ried area in Southern Hessia, Germany. In the evaluation of a new water management plan, he finds that the proposal has a significantly positive impact on the environment, whereas it would have to be rejected as inefficient on economic grounds alone.

The applications above show that there is not one single way to combine the costs and benefits of groundwater protection. As different approaches can be used to calculate costs or benefits, the results of the analysis will depend on the chosen approach. One common limitation is that all examples above focus on the optimal use of one particular instrument only, but do not try to determine the optimal protection level as such. Especially the study by Messner et al. also shows that the effectiveness of different policy measures depends largely on the reaction of behaviours and consumers. If these effects are incorporated, the complexity of the analysis increases considerably.

In terms of the different approaches that can be used to assess jointly the costs and benefits, the case study by Fuchs (1994) and Messner et al. (2001) represent extreme points: Fuchs restricts the estimation of benefits to the avoided treatment costs – and thereby considers only one of several kinds of benefits. Messner et al., by contrast, include a range of interdependent and second-round effects for both costs and benefits of groundwater protection. In addition, the case presented by Quadflieg (2002) provides an example how the incorporation of environmental effects into an economic analysis is possible without their monetisation. At the same time, while it helps to underline the trade-off between environmental and economic interests, it does not offer a rule on how they should be combined.

7.3 Examples of Risk-Based Management of Groundwater Resources

The approach to coordinate the management of groundwater contamination on a basis of risk minimisation emerged as a consequence of experiences with the US superfund programme conducted in the 1980s and 1990s (cf. chapter 4.2.1). In the course of this programme, the perception arose that large amounts of money were spent on the remediation of contaminated sites that did not actually pose a large risk for human consumption. Out of this dissatisfaction with the inefficient allocation of resources to environmental remediation, the idea developed to base the allocation of resources in such a way that the overall risks for human use were minimised, rather than eliminating all pollution throughout the country.

Essentially, risk depends on two factors: it increases with the

- *severity* of the impact and with the
- *probability* that the impact will occur.

The severity of the impact, in turn, depends on the value of the affected groundwater resource, and its vulnerability to pollution. Risk-based management, in its broadest sense, relates to policy approaches that use risk minimisation as the main criterion for the decision on a particular policy option. Consequently, risk-based management focuses groundwater protection efforts primarily on those locations where pollution would have the most severe impact, and on those areas where it is most probable that contamination will occur. Some practical applications of risk-based groundwater management are introduced in the following box.

Box 7.3: European Examples of Risk-Based Groundwater and Soil Management

Risk-based management of groundwater pollution has won widespread support by practitioners in the field (Environment Agency 1999); in Europe, it is promoted a.o. by the UK, but also by the Belgian / Flemish, Dutch and Irish authorities (OVAM 2002, Skinner 1999, Daly and Misstear 2001). So far, risk-based approaches are primarily used for dealing with point-source soil and groundwater pollution, and especially in order to decide on priorities for dealing with historically contaminated sites.

The model that is practised in ***Ireland*** comprises two elements: risk assessment and risk management. For the risk assessment, the source of contamination, the vulnerability of groundwater and the type of groundwater (aquifer, well or spring) are evaluated with a source-pathway-receptor model. After the nature and the extent of the risk have been assessed, the ensuing risk management element consists of the design and implementation of a response to the particular risk at hand (Daly and Misstear 2001).[40]

The ***Belgian / Flemish*** approach to risk-based management of contamination explicitly aims to reconcile environmental and economic objectives. The approach put forward to achieve this is referred to as *BATNEEC* (short for Best Available Technologies Not Entailing Excessive Costs) (OVAM 2002). It is used to identify the strategy with the highest financial benefit, based on the argument that if public health and the protection of the environment can be safeguarded at minimal costs, every further remediation effort will be "superfluous, time-consuming and a waste of money". Although concrete measures will have to be decided on a site-specific basis, the challenge is to unify the decision making methodology as far as possible.

In the **Netherlands**, soil protection policy has been committed to bringing soil pollution under control in 25 years.[41] The approach chosen to reach this target essentially contains three tracks: function-oriented and cost-effective remediation, use of market forces, and effective government. The Ministry Housing, Spatial Planning and the Environment (VROM) expects that the cost of soil remediation operations could be brought down by 30-50% with this approach. The new policy grew out of a general dissatisfaction with previous policies, which were perceived as ineffective and inefficient both by the public and by policymakers. To insure cost-effectiveness of soil remediation policies, their goal was shifted from complete removal of pollution to a function-oriented remediation, meaning that contamination is only reduced to a degree where the health and safety of the public, as well as the maintenance of soil functions are safeguarded

Risk-based management is not an economic approach in the strict sense, since it does not presuppose an economic valuation of the costs and benefits of different options. However, in a broader sense, it is certainly compatible with economic approaches, since it aims at allocating resources on protection and remediation in such a way that they achieve a maximum effect.

At the same time, risk-based approaches also have to incorporate economic aspects into their risk assessment: as noted above, the risk depends partly on the severity of a contamination incident, which in turn depends on the value of the affected resource. This does not necessarily have to be expressed monetarily – for example, it is possible to assess the role of unpolluted groundwater for public health without putting a monetary value on it. At the same time, many of the trade-offs and alternative uses that have to be considered when evaluating the value of an aquifer can best be captured economically.

The UK Environment Agency (1999) identifies two further ways how risk-based analyses depend on economic considerations:

- Risk-based strategies require the definition of an "acceptable level" of risk. The determination of this level should also take economic considerations into account: In relation to human health, a relevant question is whether the resources spent on groundwater protection to ensure an acceptable risk level may be spent more effectively on other, competing public health purposes
- In a purely risk-based approach, the economic attainability of a given remediation target is not considered. However, this is clearly a relevant issue, in the sense that society may deem the costs of a remediation measure as unacceptably high in relation to its benefits. It should be noted that the practical application of risk-based management, e.g. through the BATNEEC approach practised in Belgium / Flanders, aim at incorporating economic considerations as well.

[40] For a more detailed discussion, see also http://www.clarinet.at/policy/ire_approach.htm.

[41] Under the Dutch system, groundwater protection is subsumed under soil policy; for this reason it is included here.

Box 7.4: Risk-Based Assessment and Economics: the Case of Transport

Ojala (2000) has investigated the need for groundwater protection from petroleum leakage and spills of hazardous materials as a consequence of road accidents. She reports on the risk-based approach of the Swedish National Road Administration (SNRA), which combines the probability of pollution with the potential impact of pollution. In this way, a risk assessment allows estimating the socio-economic benefit of different precautionary measures, and thus enables a prioritisation of the most efficient path of action. The author remarks that, in practice, the assessment is used to order the different alternatives, rather than calculating in absolute terms the value of each measure. However, Ojala also concludes that the existing precautionary measures are not exercised efficiently in a great number of cases: the actions taken are frequently either unduly extensive, or they are not necessary given the local hydrogeological conditions.

In a comparison of ten different precautionary measures, she finds that these measures achieve risk reductions between 20 and nearly 100%. The compared measures include different impermeable membranes below the road surface, stormwater treatment plants, guard rails, kerbstones, as well as administrative measures such as information and signing measures, speed limits, emergency alerts or route guidance, i.e. the prohibition of certain roads for hazardous goods. Among the compared measures, route guidance proved to be most effective at a risk reduction of almost 100%. For some measures, there is considerable variance of up to 70 percentage points in their effectiveness. Many different measures can also be combined, which obviously increases the total risk reduction. In a next step, Ojala combines the effectiveness of different measures with their estimated costs; with costs ranging from around 10,000 € per kilometer for simple kerbstones, up to 200,000 € per kilometer for impermeable geomembranes. By and large, with the notable exception of route guidance control, she finds that more expensive measures usually achieve more risk reduction. This increases the need to design the protective measures according to local circumstances, since otherwise large sums will be invested inefficiently. On the other hand, the relatively low cost of administrative measures also means that such measures can easily be taken in addition to other measures, and at a low risk of misallocation. Therefore she recommends that high emergency alert should always be undertaken.

7.4 Economics and Target Values for Groundwater Protection

In order to assess the pollution and protection targets that are optimal from a social point of view, the standard approach in environmental economics is to perform a cost-benefit analysis (Bergman and Pugh 1997).[42] In theory, the socially optimal level is reached at the point where the marginal benefit from reducing pollution equals the marginal cost of abatement. In order to conduct such an analysis, however, estimates of the relevant cost and damage / benefit functions are needed (optimally in the form of dose-response functions), thus permitting a direct comparison of the cost and benefit of one additional unit of pollution.

In principle, such a comparison can be achieved in a full cost-benefit-analysis. Of the methods outlined above, the cost-benefit analysis is the only method that allows defining the socially optimal level of pollution *endogenously*, meaning that it is part of the output derived in the analysis. By contrast, the other methods outlined above (Cost-effectiveness analysis, Multi-criteria analyis, or Risk-based management) all rely, to a greater or lesser degree, on a predefined target level, or at least a predefined set of weights to be attached to the different decision parameters.

---

[42] An alternative, more widely used application of cost-benefit analyses is to use them to select between different discrete policy options: thereby, cost-benefit analyses are conducted for each option; in the end, the option is chosen that delivers the highest net present value. This approach is much easier to handle, since it does require the estimation of continuous cost and benefit functions; at the same time it is less informative, since it only allows the choice between policy options, but not between continuous decision parameters, such as tax levels. For this reason, it also does not deliver information on the most efficient protection target.

Unfortunately, many of the conditions required for a full economic assessment are not given in the case of groundwater pollution. Some limitations are of a general methodological kind, whereas others are due to gaps in the available knowledge of groundwater and its reaction to contamination. The following sections will first discuss some of the main limitations in methodology and data limitations; thereafter, we will offer some conclusions on what can be said despite of these limitations.

Box 7.5: Regulating Emissions or the Regulating the Environment?

Many of the limitations related to the economic assessment of groundwater protection are related to the missing link between two separate policy parameters: Emission Limit Values and Environmental Quality Standards.

Policies for groundwater protection can be targeted at two different parameters: Emission Limit Values set the maximum quantity of a pollutant that may be emitted in a certain area in a given time, whereas Environmental Quality Standards define the quality level which the policy is supposed to achieve. Consequently, emission limit values (ELVs) are easier to use for assessing the cost of a regulation, while environmental quality standards (EQSs) facilitate an assessment of its benefits.

Unfortunately, the combination of both in many cases presses the currently available knowledge beyond its bounds. Optimally, any EQS could be translated into an associated ELV, which would then allow to design an appropriate policy response. However, the link between the exposure (i.e. pollutant discharges to the aquifer) and the resulting damage (i.e. effect of these discharges) is not sufficiently clear in most cases, mainly because the needed dose-response functions are lacking and since time lags impede an efficient monitoring. The link can be calculated for a specific site and pollution problem, but the result will necessarily be dependent on the local characteristics of the individual site. Note that this is not primarily an economic problem, but is owed to limited experiences with the travelling times of pollutants and uncertainty about the involved biochemical processes:

Link between ...	*and ...*	*is provided by ...*
ELVs	costs...	economics, engineering
ELVs	EQSs	hydrogeology, biology, chemistry
EQSs	benefits	economics, biology

Some valuable new insights can be gained from computer-based modelling of agricultural land use and the associated nitrogen balance in soil and groundwater, and from the integration of these results with groundwater flow and contaminant transport and fate models (e.g. the RAUMIS or MONERIS models, or the European wide project EUROHARP to combine such models). Although these models have shown good performances in exemplary applications, they are not sufficiently developed at this stage to close the existing knowledge gaps.

However, the lack of knowledge that would be required to combine ELVs and EQSs as equivalents can be overcome by following a combined approach instead: based on the available knowledge, ELVs can be set (and policies designed accordingly) in such a way that the EQSs are reached. If the EQSs are not reached, the ELVs can be revised upwards. This argument also serves as a strong support for the continued use of command-and-control measures in groundwater protection: if there was sufficient certainty about the effect of economic instruments on groundwater-polluting activities, it would be feasible to replace command-and-control measures entirely with taxes and other economic instruments. However, since this required knowledge is clearly not given, command-and-control measures are still needed as a “safety net”. The main problem with this approach is that it blends out the sometimes extensive time lags involved; by the time the limit values have been realigned, contamination may have taken place already and may be difficult to reverse.

7.4.1 Limitations for the Use of Economics in Setting Target Values

As elaborated above, economic theory provides some insights to increasing the efficiency of groundwater protection. According to the standard economic approach, social benefits can be maximised if the target levels are set in such a way that the marginal benefit of protecting groundwater is equal to the marginal cost incurred to achieve this protection level. In practice, such a full cost-benefit analysis of groundwater protection is limited by methodological problems, and by the limited availability of economic data. As the previous chapters have shown, the information required for a full economic assessment of groundwater is far from complete. In order to evaluate where we stand in terms of a full economic assessment of groundwater protection, we will turn to methodological aspects first, and then to the limitations imposed by a lack of data.

➢ ***Methodological limitations***

Estimates of the costs and benefits of groundwater protection are always *site-specific*, reflecting the local socio-economic, hydrogeological and biophysical conditions. These limitations apply to estimates of the costs and benefits of groundwater protection alike: in both cases, it means that the transferability and completeness of findings cannot be taken for granted. In particular, this carries the following implications:

- estimates of costs and benefits that are derived from different study areas, or that apply to different types of pollutants, cannot readily be compared (cf. chapter 6.5)
- general conclusions about either the costs or benefits are limited by the fact that the chosen study areas, and the studied approaches, are not necessarily representative
- economic estimates are most reliable for human uses of groundwater. The valuation of non-human uses, i.e. ecosystem benefits, is a concern for the economic analysis, but lacks a satisfying analytical framework (cf. chapter 6.4)

A general, related problem is that the majority of case studies included in this survey focussed either on the benefits or on the costs of groundwater protection, whereas few undertook a joint assessment of both aspects. Due to the site-specific nature of most findings, and due to the different methodologies employed, it is very problematic to combine results from different regions.

A further methodological problem in defining target levels for groundwater protection arises from the fact that most empirical studies tend to focus on one pollutant in isolation. However, the need to define target levels for good groundwater status relates to a range of pollutants. If these are to be considered, a number of methodological questions need to be addressed. For example, contamination from multiple sources may require more extensive treatment due to reinforcing effects, and may reduce the scope for targeted corrective actions. At the same time, willingness to pay for protection from multiple sources may be lower than the sum of the estimates for individual pollutants, this is because averting behaviour will become more attractive as protection measures grow more complex and more costly.

With respect to assessing the ***costs*** of groundwater protection, the following methodological problems arise:

- The full costs of groundwater protection comprise indirect costs and dynamic effects as well. For example, the costs of reduced nitrate fertiliser applications go beyond the additional expenditure for less harmful fertilisation methods, or the foregone income from switching to other crops. A full assessment of the costs must also include second-order effects, such as the income losses for agriculture-related services if the income of farmers is reduced.
- An analysis of the costs of groundwater protection should also go beyond the costs of isolated policy measures, but rather turn to the optimal combination of different instruments and measures. While such an analysis is much more informative, it would also need to take reactions by different economic actors into account. This, in turn, either requires strong assumptions, or increases the research requirements significantly (cf. 4.3).

Concerning the ***benefits*** of groundwater protection, the following methodological concerns are most relevant:

- The uncertainty about the possible effects of groundwater contamination increases with the length of the time frame considered; therefore, the estimate itself will become less reliable. From the consumer's point of view, uncertainty about the extent of contamination and possible future dangers leads to increased anxiety, which can be seen as an economic cost. At the same time, the travel times and spread of contaminants become increasingly difficult to model, which reduces the accuracy of any prediction of the damage avoided through groundwater protection. Therefore, the choice of the time frame and discounting factor has a strong impact on the outcome of the analysis.
- In most analyses of the benefits of groundwater protection, the point of departure is established in very general terms only: there is typically only general information on the status quo of groundwater contamination before protection measures are implemented.
- A related point is that the proposed protection measures are typically defined very broadly as improving groundwater to a predefined "safe" level (e.g. compliance with official drinking water standards), but do not offer any alternative choices. Therefore, the available analyses of benefits are only partly suited to determine the *optimal* level: they do not estimate the benefits for protection levels other than those proposed (which may be higher or lower), and they frequently do not state by how much the situation has to be improved to reach this level.
- Finally, some complications arise from the diversity of estimation procedures that have been used to measure the benefits of groundwater protection (cf chapter 6.1). The question to which degree these procedures are complementary or substitutive has not been answered in a satisfactory way, mainly because analyses focus on one particular procedure only.

➢ ***<u>Data limitations</u>***

The limited availability of data on the costs and benefits of groundwater protection is the most pressing restriction for a full economic assessment of the costs and benefits of groundwater protection. Concerning the ***costs***, the following qualifications can be made:

- The costs of different groundwater remediation measures are difficult to compare, since the choice of technologies is mainly determined by the conditions of the specific contamination problem.
- While some research has been devoted to particular instruments, their optimal use and the costs associated with it, there is much less evidence on the combination of different instruments and the associated possible costs reductions.

For the data on the ***benefits*** of groundwater protection, the following caveats apply:

- In general, the available evidence on the benefits of groundwater protection is patchy. Bearing in mind that benefit estimation procedures are necessarily site-specific, and given the limited amount of European case studies, it is difficult to draw quantifiable general conclusions about the benefits of groundwater protection.
- Among the different estimates of the benefits of groundwater protection, there is an imbalance of evidence on selected human uses, and particularly on the use of groundwater as drinking water. Much less evidence is available on functions of groundwater more indirectly related to humans, especially its non-use values (i.e. preservation for others, or in its own right), or the ecosystem benefits of protected groundwater. However, the little available evidence indicates that these values may be significant, also in relation to the direct use-value of groundwater.
- In line with the different uses of groundwater, different procedures are used to value the benefits of groundwater protection. Not all of these are mutually compatible and lead to the same results. The cases studies based on these approaches, therefore, can just give

hints at certain values of groundwater protection. A common result is that considerable benefits arise form groundwater protection which tend to be underestimated in comparison to the financial costs of protection measures.

7.4.2 Comments on Target Values

Because of the methodological limitations mentioned above, and because of the limited empirical basis, an exhaustive discussion of economically optimal target values is not possible. To our knowledge, the question whether existing European standards for groundwater protection are economically efficient has not been addressed anywhere in the literature. Nonetheless, some comments can be offered on the applicability of economic assessments to define target values.

➢ ***Site-specificity***

The benefits and costs associated with reaching a given target value vary considerably from case to case and cannot be transferred or generalised easily: the benefits of groundwater protection will be much higher for an aquifer that is used to supply an entire city, as compared to an aquifer in a sparsely populated region. At the same time, there are common threshold values above which groundwater protection imposes costs on the regulated parties under any circumstances. A further complication is that, even for specific cases, the existing studies normally highlight a particular element of costs or benefits, seldom giving a complete comparison. Taken together, this means that the economic assessment is still limited for deriving insights other than on a site-to-site basis.

On the other hand, the fact that costs and benefits depend on site-specific circumstances implies that efficient approaches to groundwater protection will not lie in a uniform reduction of emissions: groundwater protection can be achieved most efficiently by redirecting activities with potential adverse effects on groundwater, e.g. by increasing temporal and spatial efficiency of fertiliser and pesticide applications, or by reducing polluting activities in particularly vulnerable areas.

Therefore, it appears that cost-benefit analyses cannot be used as the sole instrument for a Europe-wide assessment of the efficiency of groundwater protection. The available evidence is incomplete, and gathering the required evidence would be costly and time-consuming. Instead, cost-benefit analyses can be more effective as a tool for site-specific analyses, e.g. to assess whether temporary or spatial derogation from an otherwise binding target value or protection requirement is justified. In the context of the future EU Groundwater Directive, cost-benefit analyses, therefore, appear to be most relevant for assessing whether a derogation from the requirement to achieve good groundwater status is justified based on the cost and benefit estimates for a cost-effective set of measures.

➢ ***Imbalance between benefit and cost estimates***

As this study argued, the economic assessment of the benefits of groundwater protection is a more complex endeavour than the assessment of its costs. Because of methodological difficulties (such as the focus on groundwater used as drinking water, and the difficulties with assessing ecosystem benefits of groundwater protection), benefits are likely to be underestimated in relation to the costs. The fact that the benefits of groundwater protection are more difficult to calculate empirically does not mean that they are less tangible or less material than the costs; the problem is rather that they are harder to value economically.

➢ ***Demand for better groundwater protection***

Notwithstanding the limitations mentioned above, some important, general findings can be derived from the various studies of consumer's willingness to pay: groundwater protection is perceived as an important issue; and in many cases consumers have stated their demand for better protection, as well as a significant willingness to pay for it. While the absolute numbers of WTP cannot be generalised, the fact that WTP is significantly positive in practically all case studies reported above points to a demand for more effective protection measures. In particular, non-use values are valued as well: there is a widespread perception that groundwater resources should be preserved for future uses, even by people who do not intend to use these resources themselves. With some caution, this can be interpreted as an indication of support for the principles of non-deterioration and trend reversal, as foreseen in the Water Framework Directive and embodied in the future EU Groundwater Directive.

- ***Protection vs. Remediation***

Especially in cases of point-source pollution, numerous pollution problems arise from disposal practices that were considered as efficient and sufficiently safe at the time, but which now have emerged as insufficient, leading to high costs for the clean-up of contaminated soil and groundwater. In general, the contention is that groundwater protection is almost always cheaper than to incur pollution first and clean up later.

The past episodes of pollution also provide evidence of the evolving knowledge of the mechanisms governing groundwater contamination, and the growing concern with its protection. With the benefit of hindsight, many decisions taken in the past now appear irresponsible and short-sighted. Given the limited knowledge of the dynamics governing groundwater flows and the behaviour of contaminants, or the limited understanding of the interconnections between surface- and groundwater bodies, it is equally possible that decisions taken today may appear uninformed if viewed 40 years from now. Therefore, taking into account the precautionary principle, it is economically appropriate to give preference to protective measures over remediation, and to include a safety margin in setting target values.

8 Conclusions

Some main results of this study are presented in the following.

Methodological aspects of applying economic methods to groundwater protection

- Many instruments of an economic analysis are not easily applicable to groundwater protection. This is owed to the hydrogeological specifics of groundwater and groundwater pollution: the long and variable pollutant travel times and associated time lags between action and result; the dynamics of groundwater flows and the spread of contaminants; the potential irreversibility of pollution; the interrelation between qualitative and quantitative aspects; as well as the invisibility of groundwater for consumers are all factors that restrict the application of economic instruments straight out of the textbook (cf. chapter 2.4).

- Partly as a consequence of this, the results of an economic assessment of groundwater protection are necessarily *site-specific*. The costs and benefits of groundwater protection are largely determined by socio-economic and hydrogeological characteristics of the study area, and by the current and future uses of an aquifer. Therefore, specific statements on the economically efficient level of groundwater protection should be assessed primarily on a site-by-site basis.

- There is further scope for more *integrated research* on the economics of groundwater protection – the empirical and theoretical works surveyed in this study mainly focused on particular aspects or instruments of groundwater protection. On the side of cost assessments, more insights could be gained from research on the efficiency of different instrument combinations, as well as their integration with localised hydrogeological conditions. On the side of benefits from groundwater protection, more integration with ecosystem approaches would be particularly desirable, in order to gain a better understanding of the ecosystem benefits of groundwater protection (cf. chapters 6.4 and 6.5).

- The different *methodological approaches* to the valuation of groundwater as a non-economic good are agreed in theory, but display considerable variations in practice. While there is widespread agreement on the high value that users place on groundwater intended for consumption as drinking water, there is still some debate on less direct uses, and particularly on the value of preserving groundwater for use by future generations (cf. chapter 6.4).

- In the context of contaminated sites, *risk-based management* approaches are widely used for the choice and prioritisation of remedial sites. Such approaches offer a good first indication for the targeting of limited resources in cases where an economic assessment is not feasible. Nonetheless, risk-based assessments also rely on economic information to a certain degree (cf. chapter 7.3).

Empirical results and their assessment

- The empirical data concerning costs and benefits of groundwater protection is still fragmentary and incomplete. Therefore, a full Europe-wide assessment of the costs and benefits of groundwater protection remains a remote target. Even in pioneering countries such as the UK, where economic assessment of the costs and benefits is mandatory for all environmental policy proposals, the experience with respect to groundwater protection is still limited.

- Concerning the economic assessment of benefits from groundwater, the availability of data is more unsatisfactory than concerning the costs. Especially since assessments of the benefits have primarily focussed on the valuation of groundwater as a source of drinking water. More indirect benefit estimates, e.g. taking into account the effects of groundwater pollution on dependent ecosystems and surface water bodies, remain patchy.

- A number of studies have found that households in many regions of Europe and the US are willing to give up some income in exchange for improved groundwater protection. This finding provides strong evidence that the current situation in the respective regions is perceived as unsatisfactory. Unfortunately, the small number of European case studies does not allow any further generalisation (cf. chapter 6.3).
- There is an imbalance and lack of integration for assessment of different pollutants and kinds of pollution. Nitrate pollution has been researched fairly extensively both in terms of costs and benefits. At the other extreme, economic analyses of heavy metals contamination in groundwater could not be found. Likewise, economic assessments of groundwater protection have mainly focused on the agricultural sector, whereas much less attention has been devoted to industrial pollution (cf. chapter 4.3 and 6.5).

Policy-related results

- In recent years, a growing number of Member States have gained experiences with approaches to incorporate costs and benefits of groundwater protection measures, and to improve the targeting of resources in the field of groundwater and soil protection, e.g. through *risk-based management*. Although promising, the effectiveness of these measures awaits to be seen. There is some scope for the connection of such national approaches with the economic analysis and the selection of cost-effective sets of measures foreseen under the Water Framework Directive (cf. chapter 7.3).
- There is some scope for the extended use of economic instruments in groundwater protection. Concerning diffuse agricultural pollution, a stronger recognition of groundwater protection requirements in the ongoing agricultural subsidy reform offers itself as an alternative to introducing fertiliser or pesticide taxes. At the same time, there is evidence that sizeable reductions of agricultural emissions can be achieved at very little cost through improved agricultural management practices. Informational measures should be used to realise such *win-win*-solutions.

Overall results

This study has shown that, in general, the issues of economics and groundwater protection has so far not been investigated to a great extent. The underlying methodological limitations as well as the lack of case studies show that, in the foreseeable future, economic instruments and approaches will be more relevant for assessing the most efficient method of groundwater protection on a site-specific basis. A full, economy-wide derivation of target values and optimal solutions based solely on economic considerations is, therefore, difficult. Nonetheless, economic considerations can give some valuable insights. The following general results are of interest for establishing target values that can be derived from the present survey:

- Even where the remediation of polluted groundwater is technically feasible, it remains resource-intensive and time-consuming. New restoration technologies are developed but should not be seen as a panacea. Generally, it will almost always be less expensive to prevent groundwater contamination than to clean it up. This result is reinforced by the limited knowledge of groundwater flow dynamics and contaminant fate and transport.
- The benefits of groundwater protection are systematically under-estimated, since they include non-use values and indirect values that are difficult to assess economically. The ecosystem benefits of groundwater protection (e.g. for groundwater-dependent wetlands) can be especially significant for groundwater bodies with extensive interconnections to dependent ecosystems. However, estimates establishing the contribution of groundwater for these ecosystems remain rare.
- Because of its dependence on hydrogeological and socio-economic conditions, assessments of the costs and benefits of groundwater protection should be conducted primarily on a site-to-site basis.

- The site-specific use of economic assessments should be strengthened to support the selection of cost-effective sets of measures according to the Water Framework Directive. Their use will also be relevant in order to establish the relevant economic basis for derogations according to Article 4 of the WFD. This applies particularly to assessing the justification of time and quality derogations. The relevance of economic assessments for new modifications / activities must be decided from case to case, whereas its applicability is unlikely in the case of Heavily Modified Water Bodies.

By way of a general conclusion, it can be stated that an economic assessment can contribute some important insights to the formulation of cost-effective groundwater protection policies. However, it should not be seen as the sole instrument for the determination of groundwater protection levels. The fundamental trade-offs associated with groundwater protection, e.g. between preserving groundwater resources and current uses, are addressed in the economic literature, but are not answered conclusively.

Also in connection with the impact assessment for groundwater protection, it should be underlined that the economic assessment does not only offer insights on the costs of groundwater protection but can also contribute to assessing the (direct and indirect) economic benefits of groundwater protection. In addition, it provides ways of measuring the distributional effects associated with different policy alternatives.

While the economic assessment offers some new insights on groundwater protection, some caveats apply. The scope of an economic assessment is not only limited by scarce empirical data. A more fundamental consideration is that economic rationality is only one of several approaches that needs to be considered in the formulation of groundwater protection policies. The combination of economic views with ecological, hydrogeological and social considerations is primarily a political challenge; likewise the decision on optimal groundwater protection strategies remains a political decision.

9 References

Abdalla, C.W. (1990): Measuring economic losses from groundwater contamination: An investigation of household avoidance costs. Water Resources Bulletin 26 (3): 451-63

Abdalla, C.W. (1994): Groundwater values from avoidance cost studies: implications for policy and future research. American Journal of Agricultural Economics, 76 (5): 1062-7

Abdalla, C.W., B.A. Roach and D.J. Epp (1992): Valuing environmental quality changes using averting expenditures: an application to groundwater contamination. Land economics, 68, pp. 163-9

Agence de l'eau Adour-Garonne (2002): Test of the cost-effectiveness analysis. Summary report for the Lille conference by Stéphane Robichon, February 2002

Aguilera Klink, F., E.D. Pérez Moriana, and J. Sánchez García (1998): Environmental Valuation of Groundwater in an Island Context : the Case of Tenerife (Canary Islands), in: C3ED (1998): VALSE project report, chapter 7; see also http://alba.jrc.it/valse/documentation.htm

Archer, D.W. and J.F. Shogren (2001): Risk-induced herbicide taxes to reduce ground and surface water pollution: an integrated ecological economics evaluation. Ecological Economics 38 (2001), pp. 227-250

Association of State and Territorial Solid Waste Management Officials (ASTSWMO) (1998): Compendium of Groundwater Valuation Methodologies. April 1998

Balkau, F. (1999): Abandoned Mine Sites - Problems, Issues and Options. Paper presented at the Berlin II Roundtable on Mining and the Environment on 24 November 1999

Balmford, A., A. Bruner, P. Cooper, R. Constanza, S. Farber, R.E. Green, M. Jenkins, P. Jefferiss, V. Jessamy, J. Madden, K. Munro, N. Myers, S. Naeem, J. Paavola, M. Rayment, S. Rosendo, J. Roughgarden, K. Trumper, and R.K. Turner (2002): Economic Reasons for Conserving Wild Nature. Science, Vol. 297, 9 August 2002

Baumol, W.J. and W.E. Oates (1988): The Theory of Environmental Policy, 2nd ed. Cambridge: CUP

Bel, F., A. Lacroix, C. Le Roch, and A. Mollard (1995): Agriculture, environnement et pollution de l'eau. Grenoble: INRA/ESR

Bergstrom, J.C., K.J. Boyle and G.L. Poe (eds.) (2001): The economic value of water quality. New Horizons in Envivonmental Economics series, Cheltenham / Northampton, MA: Edward Elgar

Bergstrom, J.C., K.J. Boyle, C.A. Job and M.J. Kealy (1996): Assessing the economic benefits of ground water for environmental policy decisions. US Environmental Protection Agency Environmental Economics Report Inventory, report No. EE-0115

Boyle, K.J.; Poe, G.L.; and Bergstrom, J.C. (1994): What Do We Know about Groundwater Values? Preliminary Implications from Meta Analysis of Contingent-Valuation Studies-American-Journal-of-Agricultural-Economics; 76(5), December 1994, pp. 1055-61

Byström, O. (1998): The nitrogen abatement costs in wetlands. Ecological Economics 26 (1998), pp. 321-331

Choudhury, K., R.A. Kraemer and T. Dräger (2002): EU Enlargement – Impact of the Common Agricultural Policy (CAP) on Environment and Nature. Workshop Proceedings. BfN-Skripten 57 / 2002, Bonn: Bundesamt für Naturschutz

Coase, R.H. (1960): The Problem of Social Cost. Journal of Law and Economics, Vol. 3, pp. 1-44

Commission on Geosciences, Environment and Resources (CGER) (1997): Valuing Ground Water: Economic Concepts and Approaches (1997). Washington: National Academy Press; available at http://books.nap.edu/books/0309056403/html/index.html

Crutchfield, S.R., J.C. Cooper and D. Hellerstein (1997): Benefits of Safer Drinking Water: The Value of Nitrate Reduction. Food and Consumer Economics Division, Economics Research Service, US Department of Agriculture

Daly, D. and B.D. Misstear (2001): The Groundwater Protection Scheme in Ireland: A Risk-Based Tool For Effective Land-use Planning. Paper presented at the UNECE Second International Groundwater Workshop, Budapest, November 2001

De Haen, H. (1990): Economic Aspects of Controlling the Nitrate Contamination of Drinking Water, in R. Calvet (ed.): Nitrates-Eau-Agriculture. Versailles: INRA, pp. 23-40

Dwyer, J., D. Baldock and S. Einschütz (2000): Cross-compliance under the Common Agricultural Policy. A Report to the Department of the Environment, Transport and the Regions (DETR). London: Institute for European Environmental Policy

Ecolas (2002): Financiële behoeftenraming mbt bodemsanering. Study commissioned by the Openbare Afvalstoffenmaatschappij voor het Vlaamse Gewest (OVAM), March 2002

Environment Agency (1999): Costs and Benefits Associated with Remediation of Contaminated Groundwater: A Review of the Issues. R&D Technical Report P278 (prepared by Komex and Eftec). Bristol: Environment Agency (UK)

Environment Agency (2000): Costs and Benefits Associated with Remediation of Contaminated Groundwater: A Framework for Assessment. R&D Technical Report P279 (prepared by Komex and Eftec). Bristol: Environment Agency (UK)

Environment Agency (2002): Costs and Benefits Associated with the Remediation of Contaminated Groundwater: Application and Example. R&D Technical Report P2-078/TR (prepared by Komex and Eftec). Bristol: Environment Agency (UK)

European Environment Agency (1999): Groundwater Quality and Quantity in Europe. Copenhagen: EEA

Epp. D.J. and W. Delavan (2001): Measuring the Value of Protecting Ground Water Quality from Nitrate Contamination n Southeastern Pennsylvania, in Bergstrom, Boyle and Poe (eds.) (2001)

Fuchs, C. (1994): Kosten der Vermeidung und Entfernung von Nitrat im Grundwasser. Agrarwirtschaft Vol. 43 (2), pp. 105-115

Gardner, R.L. and R.A. Young (1998): Assessing Strategies for Control of Irrigation-Induced Salinity in the Upper Colorado River Basin, American Journal of Agricultural Economics, 70(1998), pp. 37-49

Gramel, S. and W. Urban (2001): Co-operative agreements between water supply and agriculture in Germany , in: Envrionment Agency (UK): Protecting Groundwater, conference held in Birmingham 4-5 October, 2001

Hanley, N., J. Shogren, and B. White. (1997): Environmental Economics. Theory and Practice. Oxford: Oxford University Press

Hardin, G. (1968): The Tragedy of the Commons. Science, Vol. 162, pp. 1243-8

Harrington, W., A.J. Krupnick and W.A. Spofford (1989): The Economic Losses of a Waterborne Disease Outbreak. Journal of Urban Economics, 25, pp. 116-37

Heinz, I, F. Brouwer, K. Andrews and T. Zabel (2002): Co-operative agreements in agriculture as an instrument to improve the economic efficiency and environemental effectiveness of the European Union Water Policy. Final Report of the EU project ENV4-CT98-0782. Dortmund: Institute of Environmental Research, University of Dortmund

Hellegers, P., D. Zilberman and E. van Ierland (2001): Dynamics of agricultural groundwater extraction. Ecological Economics 37 (2001), pp. 303-311

Henin, S. (ed.): Activités agricoles et qualité des eaux. Dossiers nitrates. Paris: French Ministry of Agriculture

Hofreither, M.F. and F. Sinabell (1996): Konsequenzen und Chancen einer nachhaltigen Wassernutzung durch die Landwirtschaft. Diskussionspapier Nr. 53-W-96, Februar 1996. Wien: Institut für Wirtschaft, Politik und Recht, Universität für Bodenkultur Wien.

Johnson, S.L., R.M. Adams and G.M. Perry (1991): The On-Farm Costs of Reducing Groundwater Pollution. American Journal of Agricultural Economics, 73, pp. 1063-1073

Kahlenborn, W. and A. Klaphake (2001): Recherchen und Informationsaufbereitung zu ökonomischen Instrumenten in der Grundwassernutzung. Report commissioned by the working group 'groundwater and water supply' of the German federal states' water authorities (LAWA). Berlin: Ecologic

Kim, C.S., G.D. Schaible and S.G. Daberkow (2000): An efficient cost-sharing program to reduce nonpoint-source contamination: theory and an application to groundwater contamination. Environmental Geology Vol. 39(6), April 2000, pp. 649-59

Kraemer, R.A. and K.M. Banholzer (1999): Tradable Permits in Water Resource Management and Water Pollution Control, in: Organization of Economic Co-operation and Development (OECD) (ed.): Implementing Domestic Tradable Permits for Environmental Protection, pp. 75-107. Proceedings. Paris: OECD.

Kraemer, R.A., E. Interwies and E. Kampa (2002): Tradable Permits in Waterresource Protection and Management - A Review of Experience and Lessons Learned, in: OECD (ed.): Domestic Tradable Permits: Lessons and Future Directions. Paris: OECD.

Küchli, C. and B. Meylan (2002): Wälder liefern das beste Trinkwasser. Wald und Holz Vol. 10/02, pp. 51-54

Lacroix, A. and F. Balduchi (1995): Le traitement des nitrates de l'eau potable. Bilan économique et perspectives. Document de travail, INRA/ESR, Grenoble, (94-06)

Lee, I. and E.G. Nielsen (1987): The Extent and Costs of Groundwater Contamination by Agriculture. Journal of Soil and Water Conservation, 42, No. 4, 243-48

Lindahl, E. (1919): Die Gerechtigkeit der Besteurung, 1919. (translated. as "Just Taxation: A positive solution", 1958)

Massarutto, A. (1999): Agriculture, Water Resources and Water Policies in Italy, Fondazione Eni Enrico Mattei Note di Lavoro: 33/99, May 1999, pages 24.

Messner, F., B. Klauer and M. Drechsler (2001): Der Nutzungskonflikt zwischen Trinkwasserschutz und wirtschaftlicher Entwicklung im Torgauer Raum, in F. Beckenbach et al. (eds.) (2001): Ökonomische Naturbewertung. Jahrbuch Ökonomische Ökonomik, Bd. 2. Marburg: Metropolis

Miljøstyrelsen (2002): Metoder til værdisætning af dansk vejtrafiks forurening af jord og grundvand (Methods for the valuation of soil and groundwater pollution from Danish road traffic), Miljøprojekt Nr. 712 2002; english summary available at http://www.mst.dk/udgiv/publikationer/2002/87-7972-204-0/html/samfat_eng.htm

Ministerie van Volkshuisvestiging, Ruimtelijke Orde en Milieu (1999): Interdepartmental policy review: soil remediation; Letter from the minister containing the government statement relating to the function-oriented and cost-effective approach to soil pollution. The Hague: VROM

Moreaux, M. and A. Reynaud (2002): Optimal management of a coastal aquifer under saline intrusion, paper presented at the Canadian Resource and Environmental Economics Study Group 12th Annual Meeting, Montreal, 19-10-2002

Organisation for Economic Cooperation and Development (OECD) (2002): Environmentally Related Taxes database; http://www.oecd.org/env/policies/taxes/

Office for Technology Assessment (1985): Superfund Strategy. Washington, DC: U.S. Congress, Office of Technology Assessment, OTA-ITE-252, April 1985.

Ojala, L. (2000): Water source protection – a compilation and evaluation of the Swedish National Road Administration's work, Master thesis, Quaternary geology / Hydrogeology, Department of Earth Sciences, Uppsala University. (Original in Swedish)

Olson, M. (1965): The Logic of Collective Action: Public Goods and the Theory of Groups. Cambridge, MA: Harvard University Press

Openbare Afvalstoffenmaatschappij voor het Vlaamse Gewest (OVAM) (2002): Afweging bodemsaneringsvarianten. Draft version 24/09/02

Pfaffenberger, W., U. Scheele (1990): Kosten des Grundwasserschutzes: Eine empirische Analyse für die Bundesrepublik Deutschland. Zeitschrift für Umweltpolitik und Umweltrecht, Vol. 13 No. 2, pp. 157 – 78

Pigou, A.C. (1920): The Economics of Welfare.

Poe, G.L. and R.C. Bishop (1999): Valuing the Benefits of Groundwater Protection when Exposure Levels are Known. Environmental and Resource Economics, 13, pp. 341-67

Poe, G.L. and R.C. Bishop (2001): Information and the Valuation of Nitrates in Ground Water, Portage County, Wisconsin, in Bergstrom, Boyle and Poe (eds.) (2001)

Poe, G.L., K.J. Boyle and J.C. Bergstrom (2001): A preliminary meta analysis of contingent values for ground water quality revisited, in Bergstrom, Boyle and Poe (eds.) (2001)

Press, J. and T. Söderqvist (1998): On Estimating the Benefits of Groundwater Protection: A Contingent Valuation Study in Milan, in T.M. Swanson and M. Vighi: Regulating Chemical Accumulation in the Environment, pp. 121-182. Cambridge: CUP (also published as Fondazione Eni Enrico Mattei Note di Lavoro: 53/96, July 1996, reprinted as No. 107 of the Beijer Reprint Series)

Raucher, R.L. (1983): A conceptual framework for measuring the benefits of groundwater protection. Water Resources Research, Vol. 19 No. 2, pp. 320-6

Skinner, A. (1999):Groundwater Protection – An Overview of Current Issues. Proceedings of the Seminar “Groundwater Protection in Ireland”, Enniskillen, 8 October 1999. Available at http://www.nwl.ac.uk/gwf/proceed.pdf

Söderqvist, Tore (1998): Empirical cost equations for wetland creation: The case of wetlands as nitrogen sinks in Scania, South Sweden. Beijer Discussion Paper No. 118, Stockholm: Beijer International Institute of Ecological Economics

Stenger-Letheux, A., and M. Willinger (1998): Preservation value for groundwater quality in a large aquifer : a contingent valuation study of the Alsatian aquifer, *Journal of Environmental Management*, 53, pp.177-193.

Stiglitz, J. (2000): Economics of the Public Sector, 3rd ed. WW Norton & Co

Strosser, P., M. Pau Vall and E. Plötscher (2002): Water and agriculture: contribution to an analysis of a critical but difficult relationship http://europa.eu.int/comm/agriculture/envir/report/en/eau_en/report.htm

Sun, H.; J.C. Bergstrom and J.H. Dorfman (1992): Estimating the Benefits of Groundwater Contamination Control. Southern Journal of Agricultural Economics; 24(2), December 1992, pp. 63-71

Tervonen, J., E. Alasaarcla and R. Svento (1994): Household Water Quality and Consumer Welfare: An Application to the City of Oulu, Aqua Fennica 24,1, 83-92

Tietenberg, T. (1992): Environmental and Natural Resource Economics, 3rd ed. New York: Harper Collins

Traoré, N., N. Amara and R. Landry (1999): Households' Response to Groundwater Quality Degradation. Results from a Household Survey in Quebec. Cahiers d'économie et sociologie rurales, n° 52, 1999 Cahiers d'économie et sociologie rurales, n° 52

Turner, R. K.; J.C.J.M. van den Bergh, T. Söderqvist, A. Barendregt, J. van der Straaten, E. Maltby and E.C. van Ierland: Ecological-economic analysis of wetlands: scientific integration for management and policy. Ecological Economics 35 (2000), pp. 7-23

United States Environmental Protection Agency (US EPA) (1999): EPA Groundwater Cleanup: Overview of Operating Experience at 28 Sites. Report No. EPA 542-R-99-006, September 1999. Washington: EPA, Solid Waste and Emergency Response

Victoria Transport Policy Institute (2002): Transportation Cost and Benefit Analysis - Techniques, Estimates and Implications; esp. Ch. 5.15: Water Pollution and Hydrologic Impacts. Victoria, BC: VTPI. Available online at http://www.vtpi.org/tca/

Weitzmann, M.L. (1974): Prices vs. Quantities. Review of Economic Studies, Vol. XLI (10/1974), pp. 477-94

Whitehead, J.C. and G. Van Houten (1997): Methods for Valuing the Benefits of the Safe Drinking Water Act: Review and Assessment.

Whitehead, J.C., T.J. Hoban and G. Van Houten (1998): Willingness to Pay and Drinking Water Quality: An Examination of the Averting Behavior Approach. Paper presented at the 68th annual conference of the Southern Economic Association in Baltimore, MD, November 1998.

10 Annex: Overview of Empirical Case Studies

10.1 Studies on the Cost of Groundwater Protection and Remediation

Study	Study Region	Method used	Factors assessed	Results	Remarks
Ojala 2000	Sweden	Market prices	Cost of groundwater protection measures from transport-related sources	€ 10,000 - 200,000 / km (various measures)	
Messner 2002	Torgau, Germany	Scenario-based multi-criteria analysis	Cost of protection compared to reference scenario (economic development)	-€ 9,9 m to -€ 9,5 m	*Negative costs* of groundwater protection due to existing oversupply in the region
Ecolas 2002	Belgium	Market prices	Clean-up cost for contaminated sites	€ 600,000 / site (average)	60% of costs below € 100,000 Costs up to € 45 m per site
Pfaffenberger 1990	Germany	Hypothetical calculation based on market prices	Cost of land acquisition in catchment areas	€ 465 billion investment € 2.5 bill. running cost / y (= 50 ¢ / m^3 / y)	Hypothetical calculation, assuming large-scale afforestation of catchment areas
Heinz et al. 2002	Nine regions in DK, F, D, NL, UK	Market prices	Cost of cooperative agreements in different European regions	€ 200,000 / y (average)	Costs ranging from € 9,300 to € 960,000
Gramel & Urban 2001	Hesse, Germany	Market prices	Cost of cooperative agreements for nitrate reduction	10 ¢ - 29 ¢ / m^3 of groundwater abstracted	
US EPA 1999	USA	Market prices	Cost of 28 clean-up measures	US$1.9 million investment cost per site (average) US$190,000 running cost per site per year	
Rejesus and Hornbacker 1999	US (various regions)	Market prices / reduced crop yield	Crop yield loss from reduced fertiliser applications along with improved fertiliser management	No additional cost	Only costs for farmers – cost from information provision not included
Johnson et al. 1991	Columbia Basin, Oregon, US	Market prices / reduced crop yield	Costs of uniform reduction of fertiliser applications	Crop yield loss: 10 - 22%	No monetary cost estimates available

10.2 Studies on the Benefits of Groundwater Protection and Remediation

Study	Study Region	Method used	Factors assessed	Results	Remarks
Kosz 1996	Danube floodplains, Vienna	Willingness-to-pay Avoided treatment cost	Value of wetlands for groundwater	€ 44 - € 105 million	
Küchli & Meylan 2002	Switzerland	Avoided treatment cost	Value of forests for groundwater protection	€ 54 million / year	Only use as drinking water considered
Lacroix & Balduchi 1994	France	Avoided treatment cost	Cost for nitrate treatment in 25 plants in various regions	24 ¢ - 28 ¢ / m^3 € 19 - 22 / inhabitant / y	Only use as drinking water Considered
Hofreither and Sinabell 1996	Austria	Avoided treatment cost	Drinking water purification costs for municipal water suppliers	€ 205 - 214 m investment € 22 - 39 m running cost	80% of costs for nitrate treatment Only use as drinking water considered
Gramel & Urban 2001	Hesse, Germany	Avoided treatment cost	Benefits of cooperative agreements	25 ¢ - 75 ¢ / m^3 of groundwater abstracted	Only use as drinking water considered
Traoré et al. 1998	Quebec, Canada	Averting behaviour	Expenses for bottled water	€ 100 - 145 / person / y	Only use as drinking water considered
Abdalla 1990	Pennsylvania, US	Averting behaviour	Expenses for bottled water and home treatment devices	US$ 123 - 252 / household / y	Only use as drinking water considered
Tervonen 1994	Oulu, Finland	Willingness-to-pay	Valuation of groundwater as a source of drinking water	€ 54 / household / year	Only use as drinking water considered
Stenger & Willinger 1994	Alsace, France	Willingness-to-pay	Valuation of groundwater protection	€ 98 / household / year	
Press & Söderqvist 1998	Milan, Italy	Willingness-to-pay	Valuation of reduced atrazine concentrations in groundwater	€ 425 – 559 / household / year	
Sun, Bergstrom and Dorfman 1992	Dougherty County, GA, US	Willingness-to-pay	Reduced groundwater pollution from agricultural chemicals	US$ 641 / household / y	
Epp and Delavan 2001	Pennsylvania, US	Willingness-to-pay	Government plan to reduce groundwater contamination	US$ 648 – 888 / household / y	
Crutchfield et al. 1997	Four regions in the US	Willingness to pay	Reduced nitrate exposure in drinking water	US$ 540 - 720 / household / y	Only use as drinking water considered
Edwards 1988	Cape Cod, Massachusetts, US	Willingness-to-pay	Aquifer management plan to reduce the probability of nitrate contamination	US$ 1860 / household / y	
Schultz and Lindsay 1990	Dover, New Hampshire, US	Willingness-to-pay	Plans to protect community groundwater supplies	US$ 180 / household / y	

Sun, Bergstrom, and Dorfman 1992	Doughtery County, Georgia, US	Willingness-to-pay	Protecting "safe" groundwater from potential future contamination	US$ 804 / household / y	
McClelland, et al. 1992	National (US)	Willingness-to-pay	Complete groundwater cleanup from a 40% contamination	US$ 144 / household / y	
Caudill and Hoehn 1992	Michigan, US	Willingness-to-pay	Action to prevent contamination; maintenance of well water quality	US$ 780 / household / y	
Poe and Bishop 1992	Portage County, Wisconsin, US	Willingness-to-pay	Groundwater protection program to prevent nitrate contamination	US$ 288 / household / y	
Jordan and Elnagheeb 1993	Georgia, US	Willingness-to-pay	Preventing groundwater pollution that would make sure water is safe for drinking	US$ 168 - 192 / household / y	
Powell, Allee, and McClintock 1994	Massachusetts, New York, and Pennsylvania, US	Willingness-to-pay	Establish water supply protection districts that would ensure safe drinking water	US$ 72 / household / y	Only use as drinking water considered
Laughland, et al. 1996	Milesburg, Pennsylvania, US	Willingness-to-pay	Connection to an alternative source so that drinking water meets standards	US$ 276 / household / y	
Clemons, Collins, and Green 1995	Martinsburg, West Virginia, US	Willingness-to-pay	Wellhead protection program to eliminate risk of contamination	US$ 21,6 / household / y	
Krug 1995	Western Massachusetts, US	Willingness-to-pay	Aquifer Protection District and purchase of a private water filter	US$ 84 / household / y	
Power et al. 1991	12 communities, north-eastern USA	Willingness-to-pay	Increase water supply protection	US$ 738 / household / y	
Kwak and Russel 1994	Seoul, South Korea	Willingness-to-pay	Government plan to reduce probability of major contamination incidents to near zero	US$ 480 / household / y	
Folke 1991	Gotland, Sweden	Market prices	Wetland services: nutrient sink, fish, water supply	US$ 240 / ha	No direct reference to groundwater
Hanley and Craig 1991	Scotland	Willingness-to-pay	Wetland value: use, non-use	US$ 34 / ha	No direct reference to groundwater
Tomasin 1991	Po delta, Italy	Willingness-to-pay	Wetland services: fish, hunting, recreation	US$ 1,500 / ha	No direct reference to groundwater

Kosz et al. 1991	Vienna, Austria	Market prices Willingness-to-pay	Wetland services: forest production, grassland, fish, recreation	US$ > 522 / ha	No direct reference to groundwater
Bateman et al. 1993	East Anglia, UK	Willingness-to-pay	Wetland services: recreation	US$ 108 - 266 / person	No direct reference to groundwater
Ecotec 1993	Aquatic ecosystems, UK	Willingness-to-pay	Wetland value: use, non-use	US$ 42 / household	No direct reference to groundwater
Gren 1993	Stockholm, Sweden	Avoided treatment cost	Wetland services: nitrogen sink	US$ 430 / ha	No direct reference to groundwater
Gravener 1994	Oxelösund, Sweden	Willingness-to-pay	Wetland value: use, non-use	US$ 570-1,150 / ha US$ 1.3-585 / person	No direct reference to groundwater
Gren 1994a	Gotland, Sweden	Market prices	Wetland services: nitrogen sink, fish, water supply	US$ 239-585 / ha	No direct reference to groundwater
Gren 1994b	Danube (floodplains)	Market prices Willingness-to-pay	Wetland services: N sink, forest production, hunting, fish, grassland, recreation	US$ 458 / ha	No direct reference to groundwater
Ungerman 1994	Nové Mlyny, Czech Republic	Market prices Willingness-to-pay	Wetland services: forest and agriculture, hunting	US$ 290 / ha	No direct reference to groundwater