Groundwater Management for Agriculture and Nature: an Economic Analysis

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Abstract

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As a result of declining groundwater levels, nature in the Netherlands is suffering from desiccation. Since measures taken to raise groundwater levels in order to restore nature often lead to unintended wet damage to crops in adjacent farmland, an economic analysis to determine optimal solutions is required. The fundamental factors involved in such an analysis are presented in this thesis. The main objectives are: (1) to gain an economic insight into conflicting interests between agriculture and nature with respect to the desiccation of nature; (2) to develop methods and models to analyse groundwater level management; (3) to study agricultural groundwater extraction; and (4) to provide an insight into the suitability of policy instruments for both groundwater level and groundwater extraction management.

Cost-benefit analysis has been applied to study optimal uniform groundwater level management in agricultural areas with special ecological value. Cost-effectiveness analysis has been used to study changes in non-uniform groundwater levels in nature reserves and adjacent agricultural areas. Optimal control theory has been used to study the dynamics of agricultural groundwater extraction management. The novelty of this study is the integrated economic and ecohydrological approach to the desiccation problem and the development of a method to compare various objective functions of nature restoration.

The study shows that the failure of markets, institutions and policies has resulted in the desiccation of nature in the Netherlands. Markets fail due to the public nature of groundwater services and to the externalities of groundwater extraction. There is institutional failure since it is not clear who owns rights to lower groundwater levels and to extract groundwater. Besides, water boards make decisions about surface water levels, which directly affect groundwater levels, although they are not responsible for groundwater level management. Policy reform is, however, usually conditional upon the size of efficiency gains relative to transaction costs. The study shows that the current policy goal to reduce the desiccated surface area will not maximise the increase in the ecological value. Since the marginal costs to agriculture per conservation value unit restored increase, the partial restoration of many reserves rather than the full restoration of a few reserves could be considered. The study also shows that the annual costs of damage to agriculture due to restoration are small compared to the annual costs of investments in hydrological measures.

Key words: desiccation of nature, economics of water management, groundwater extraction, groundwater level management, ecohydrology, agriculture, policy instruments.

Voorwoord

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Contents

1 General introduction

1.1 Problems of water management

In recent years, there has been a growing recognition that managing water as an economic good is an important way of achieving the efficient use of water resources, as highlighted at several conferences. The Second World Water Forum (The Hague, March 2000) stressed that decisions on water allocation among competing uses require a better analysis of the value of water (SWWF, 2000). The International Conference on Water and the Environment (Dublin, January 1992) emphasised that failure to recognise the economic value of water has led to environmentally damaging uses of the resource (ICWE, 1992). The United Nations Conference on Environment and Development (Rio de Janeiro, June 1992) underlined that the role of water as a social and economic good should be reflected in demand management mechanisms (UNCED, 1992). International bodies (OECD, 1989; World Bank, 1993; FAO, 1994) also recognise the need to study the efficient allocation and management of water.

Groundwater management is an important issue in the Netherlands, where nature is largely adapted to wet and moist environmental conditions, as a result of shallow groundwater levels. However, during the second half of the twentieth century, groundwater levels decreased, mainly due to the intensified drainage of farmland and steadily increasing extraction of groundwater. As a result of declining groundwater levels, nature in the Netherlands is suffering from desiccation. (In Dutch, this major environmental problem is called *verdroging*.) In the Netherlands in the year 2000, about 490,000 hectares of land (=14% of the total surface area of 35,000 km^2) were suffering from desiccation. The government aims to reduce the area of desiccated land by at least 40% by the year 2010 compared to the situation in 1985, when 600,000 hectares were desiccated. Since measures taken to raise groundwater levels in order to restore nature often lead to unintended wet damage to farmland, economic analyses to determine optimal solutions are required.

In this thesis, the desiccation of nature in the Netherlands is analysed using an integrated economic ecohydrological approach. In order to make the right choices about groundwater management; it is essential to improve the understanding of relationships between the hydrological, economic and institutional systems. The multidisciplinary approach used in this thesis contributes to the comprehension of these relationships as well as to the understanding of the technical, economic and institutional causes of the desiccation problem.

1.2 Hydrological conditions of the Netherlands

The Netherlands may be divided into a relatively low and a relatively high part: the west and the north are low, whereas the east and the south are relatively high (Figure 1.1). It is generally more difficult to control the water level in the high part than in the low part, which mainly consists of Holocene clay and peat and predominately consists of polders, in which the water level is artificially controlled. A polder is constructed by isolating land from the surrounding hydrological regime by means of dikes. In the western part of the Netherlands, the deep groundwater is brackish or saline except under the coastal dunes. The high eastern part mainly consists of Pleistocene cover sands. Groundwater levels in this part are hard to control, since drainage is mostly brought about by gravity (Witte, 1998). For water conservation purposes, the small rivers and brooks are provided with adjustable weirs. The groundwater is fresh and used for public water supply and sprinkling.

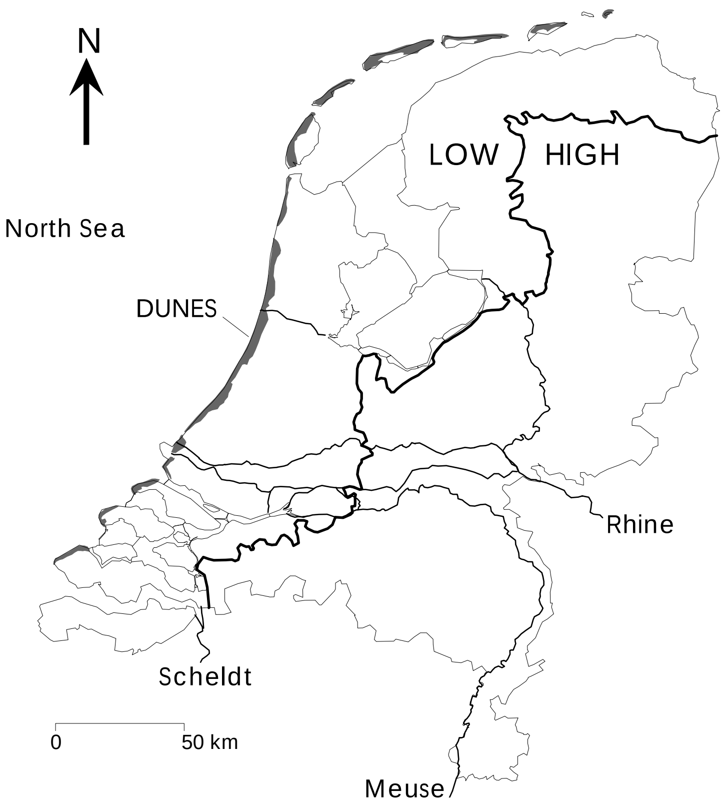

Figure 1.1 Division of the Netherlands into a western and northern low part, and an eastern and southern high part. One metre above mean sea level was chosen as the critical value of the boundary between the two parts. Source: Witte (1998).

Groundwater tables in the Netherlands are rather shallow, because of the country's low lying soil surface. A distinction is made between groundwater level and groundwater depth. Level is the height of the groundwater table relative to the surface water level of the farmland, and depth is the height of the groundwater table relative to the soil surface. For instance, in 90% of the country the groundwater depth in winter is less than 1 m below soil surface (bss) and in summer less than 2.5 m bss (Colenbrander et al., 1989). As a consequence, conservation values in the Netherlands are mainly associated with wet and moist conditions, such as fens, bogs, wet heathlands, swamp woodlands and wet meadows. Most of these ecosystems are restricted to small nature reserves (often less than 1 km^2) or spots within an agricultural landscape. As a result, these ecosystems are strongly dependent on the water management in adjacent farmland (Witte, 1998). The ecological values of areas decrease since rare plant species adapted to wet and moist environments disappear due to declining groundwater levels, which is an important cause of deterioration of nature in the Netherlands.

In the second half of the twentieth century, human interventions caused the water regime of most areas to change drastically. Since 1950 several hundred land improvement plans have been realised, and many involved the radical revision of the water management. Moreover, groundwater extraction has increased. As a result, groundwater levels have dropped, especially in the high part of the Netherlands. Feddes et al. (1999) estimate that in this area the groundwater level has dropped by an average of 60 cm. About 50% of the structural lowering of the groundwater level can be attributed to the increased drainage of farmland, 30% to groundwater extraction for public and industrial purposes, and 20% to other causes, such as changes in land use, sprinkling, urbanisation and increase in the evapotranspiration.

The intensification of agricultural activities has contributed to the lowering of groundwater levels in several ways, i.e. through the intensified drainage of agricultural areas in order to raise production, increased agricultural groundwater extraction for sprinkling purposes, and the enhanced evapotranspiration of crops, as a result of increased crop yields (Dufour, 2000).

It is important to realise that both crop growth and nature development depend on the availability of soil moisture in the root zone. Drainage and agricultural groundwater extraction do, however, indirectly affect the availability of soil moisture in the root zone. They determine, in the case of shallow groundwater levels, the processes of capillary rise and percolation (see Figure 1.2). On the other hand irrigation water directly affects the availability of soil moisture. The water balances of the unsaturated zone, the saturated zone as well as both zones combined are schematically represented in Figure 1.2.

The change in water storage of the unsaturated zone ΔS_{un} is the difference between the in- and outgoing flows, i.e. ΔS_{un}= IN–OUT. Changes in water storage of the saturated zone ΔS_{sa} and that of the combined unsaturated-saturated zone ΔS_{to} can be formulated in the same way.

$$\text{Unsaturated zone:}\quad \Delta S_{un} = \text{IN-OUT} = I + E + G - J - O \qquad (1.1)$$

$$\text{Saturated zone:}\quad \Delta S_{sa} = \text{IN-OUT} = -E - G + O + U_{in} - U_{out} + V_{in} - V_{out} \qquad (1.2)$$

$$\text{Total balance:}\quad \Delta S_{to} = \text{IN-OUT} = I - J + U_{in} - U_{out} + V_{in} - V_{out} \qquad (1.3)$$

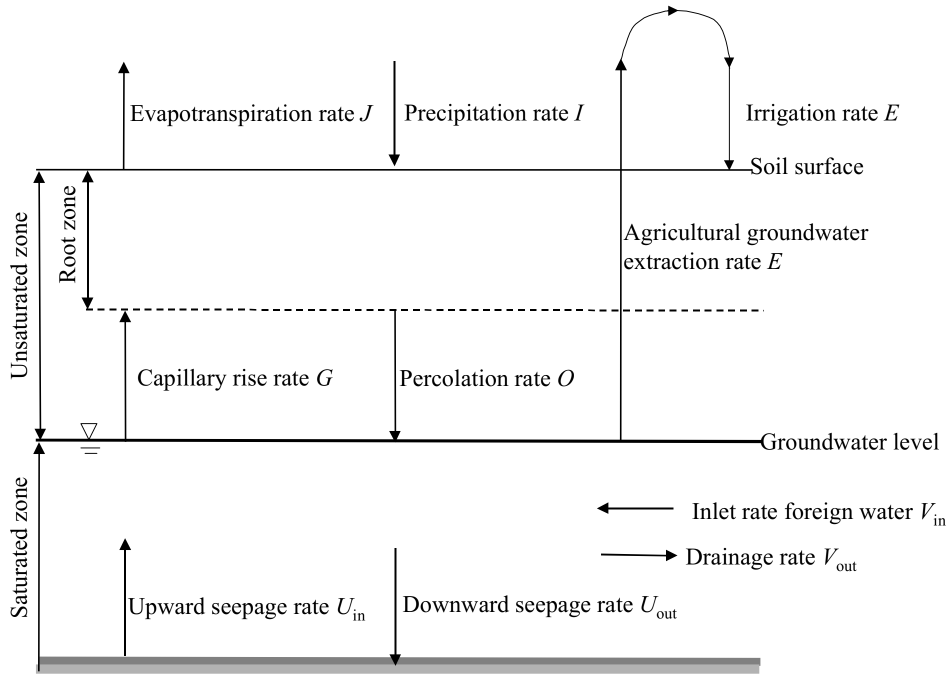

Figure 1.2 Schematic representation of water flows in the unsaturated and the saturated zone. Agricultural groundwater extraction E is applied as irrigation water. It is important to note that the symbols applied are not commonly used in hydrology, because many symbols are used in this thesis for economic nomenclature.

1.3 Desiccation of nature in the Netherlands

The main effects of hydrological changes on important operational factors are shown in a simplified way in Figure 1.3 (cf. Witte et al., 2001). First of all, a declining groundwater level may lead to less soil moisture being available to the vegetation and, as a result, in the physiological desiccation of the vegetation. Species adapted to wet and moist environments will disappear. Secondly, a declining groundwater level may cause increased aeration, which promotes mineralisation and consequently eutrophication. Hence, species characteristic of nutrient-poor sites will disappear. When organic matter is mineralised, protons are released and acidification of the soil takes place, causing species of neutral and alkaline sites to vanish. If the soil was originally influenced by upward seepage flow, a declining groundwater level may enhance eutrophication and acidification. Of course, both effects may also take place if the intensity of upward seepage diminishes. Finally, the inlet of foreign water may lead to eutrophication, since this water is often rich in phosphorus and nitrogen. Even if nutrient concentrations are low, the inlet-water may stimulate mineralisation. If foreign water is used to counteract low groundwater levels, salinisation might occur leading to the disappearance of fresh-water species.

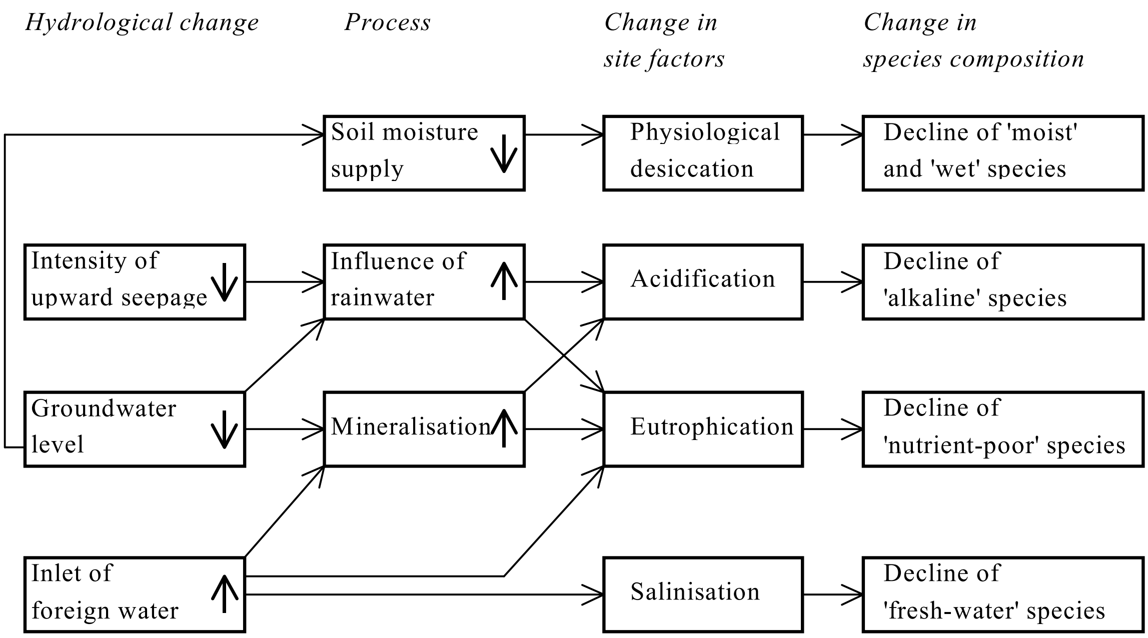

Figure 1.3 Main effects of hydrological changes on the environment. Direction of change indicated by arrows: (↑) increase, (↓) decrease.
Source: Witte et al. (2001).

Runhaar (1999) provides insight into the effects of hydrological changes on ecosystem types in the Netherlands. Oligotrophic and mesotrophic wet ecosystems (especially bogs, wet heaths, wet forests and hay meadows), which are more common in the high, sandy part of the Netherlands, are the most affected. Eutrophic wet ecosystems, which are more common in the low part of the Netherlands, are relatively less damaged.

In this study, desiccation is restricted to the deterioration of nature due to declining groundwater levels, although in the definition of desiccation in general the deterioration of nature due to decreasing intensities of upward seepage and to the inlet of foreign surface water is also considered. Figure 1.3 shows that even in the narrow definition used in this study, desiccation is more than solely physiological desiccation. It also includes eutrophication and acidification due to declining groundwater levels. In fact, the influence of their changes on plant species is thought to be of more importance than the influence of physiological desiccation (Witte, 1998).

A nation-wide investigation into the loss of conservation values since 1950 showed moderate to severe damage to ecosystems in 50% of the groundwater-dependent reserves. Figure 1.4 shows the average degree to which nature conservation areas are affected.

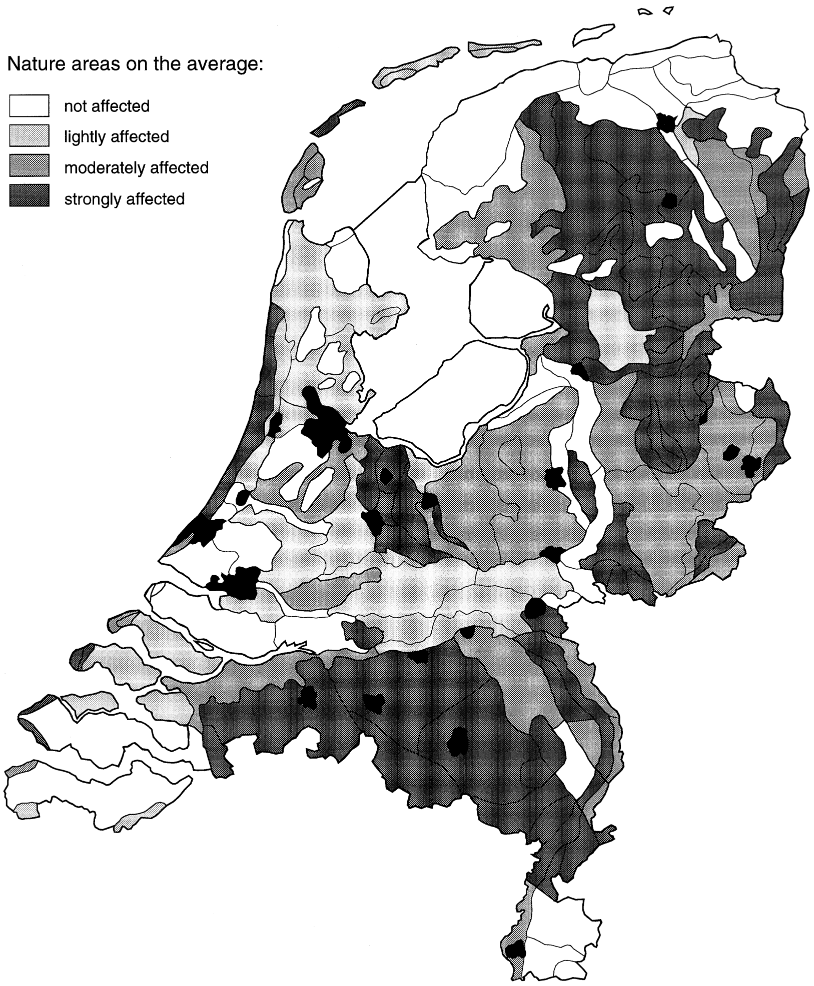

Figure 1.4 The average degree to which nature reserves have been damaged as a result of desiccation per ecohydrological region.
Source: van Amstel et al. (1989).

The major causes of the desiccation of nature in the low part of the Netherlands and those in the high part differ. The desiccation of nature due to decreasing intensities of upward seepage and to the inlet of foreign surface water is concentrated in the low part, whereas desiccation due to declining groundwater levels mainly occurs in the high part and the coastal dunes. The desiccation of nature due to declining groundwater levels in the low part is less prominent, since here the inlet of foreign surface water can easily be realised. This is contrary to the situation in the high part, where the sloping land means that the inlet of foreign surface water is both complicated and costly. In the coastal dunes, large quantities of fresh water are extracted for public water supply. The aquifer in this area would have been depleted long ago were it not for artificial recharge with river water.

1.4 Objectives of the study

The objectives of this study are fourfold:

- *To gain an economic insight into conflicting interests between agriculture and nature with respect to the desiccation of nature in the Netherlands;*
- *To develop methods and models to analyse groundwater level management;*
- *To study agricultural groundwater extraction;*
- *To provide an insight into the suitability of policy instruments for both groundwater level and groundwater extraction management.*

To achieve these objectives, the following main research questions will be addressed:

- What are the economic and institutional causes of desiccation of nature in the Netherlands?
- How can optimal groundwater level management be analysed?
- How can agricultural groundwater extraction management be analysed?
- What kind of policy reforms for groundwater management can be recommended?

In this thesis insight will be provided into the underlying economics of water management. Micro-economic analysis techniques, such as cost-benefit analysis, cost-effectiveness analysis and optimal control theory, will be applied to study groundwater management.

Groundwater analyses are often done in a stochastic framework because the availability of shallow groundwater varies over time and space depending on weather conditions. Although it is a challenge to study risk management of such variability, this thesis does not focus on this particular problem, since the desiccation of nature is mainly the result of the structural lowering of groundwater levels.

1.5 Conceptual framework

A conceptual framework that integrates hydrological, economic and institutional aspects of groundwater management is shown in Figure 1.5. This integrated framework is helpful for analysing water problems and for making the right choices about groundwater management. The framework consists of three modules (see part A in Figure 1.5):

- The *hydrological* module simulates the impact of actual and intended human interventions in the water system on the characteristic variables of the water system: groundwater availability, groundwater quality and groundwater depth. It takes account of site-specific characteristics, such as geohydrological properties.

- The *economic* module evaluates the impact of changes in the characteristic variables of the water system on the costs and benefits to agriculture and nature.

- The *policy* module reviews alternative options for the allocation of water, distribution of costs and benefits, and policy instruments to be used. These policy instruments, in turn, affect human interventions.

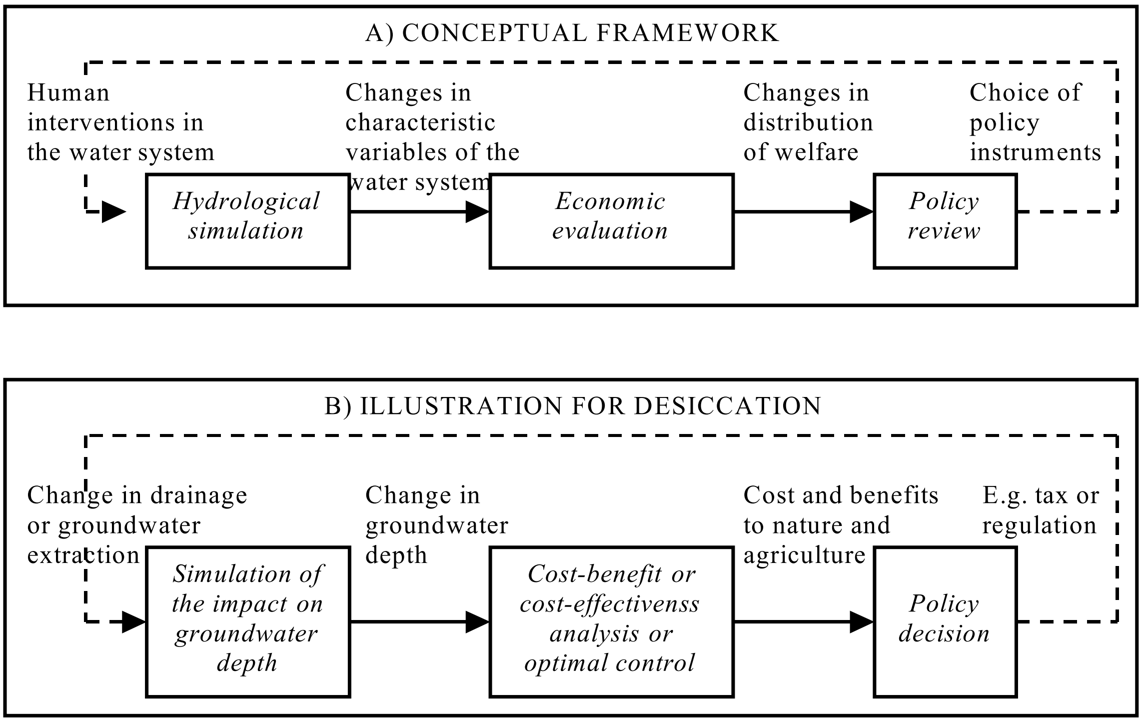

Figure 1.5 Conceptual framework to analyse water problems (A) and its illustration for desiccation (B). A block represents a module. An arrow represents input to/output from a module. The dotted arrow represents the impact of policy instruments on human interventions in the water system.

The framework is used to analyse the conflicting interests between agriculture and nature with respect to drainage and agricultural groundwater extraction (see part B in Figure 1.5). Changes in drainage or groundwater extraction in agricultural areas may affect groundwater depth in adjacent nature reserves. A detailed hydrological analysis is used to show how groundwater depth develops in the farmland and nature reserves. Once the impact of changes in drainage and groundwater extraction on groundwater depth is assessed, cost-benefit analysis, cost-effectiveness analysis or optimal control theory is applied to provide insight into costs and benefits to agriculture and nature due to changes in groundwater depth.

Cost-benefit analyses have been widely used for evaluating investments (project appraisal) and policies in the water resources area to determine whether projects or policies will be worthwhile (e.g. Howe, 1971; Sinha and Bhatia, 1982; Hanley and Spash, 1993). According to the FAO (1994) cost-benefit analysis practice needs to be modified and extended in several ways in the context of water policy review. Firstly, it should be applied to both supply-augmenting and demand-management measures. Secondly, the efficiency criterion in cost-benefit analysis should be supplemented by other criteria such as equity[1]. Thirdly, both costs and benefits should include economic estimates of the monetary values of environmental impacts.

Cost-effectiveness analysis is applicable where monetary benefits cannot be adequately measured. It is useful in comparing alternative ways of attaining a given level of benefits, i.e. it can be used to minimise the costs of achieving predefined environmental targets (e.g. Dellink and van der Woerd, 1997; Schmieman and van Ierland, 1999; Schmieman, 2001). It can also be used to allocate a given budget in the most effective way, i.e. to maximise environmental targets. It is less suitable for determining whether an objective is worthwhile.

Optimal control theory, using the maximum principle, is a technique for solving constrained dynamic optimisation problems (see Conrad and Clark, 1987; Chiang, 1992; Perman et al., 1999). Burt (1964 and 1975) contributed fundamentally to the understanding of optimal water management in a dynamic setting. He incorporates both the direct and the external costs of extraction and the trade-offs between diminishing returns to water uses now and the present value of future uses into a dynamic optimisation framework. More recently, optimal control theory has also been used by others to compare optimal groundwater extraction and open access outcomes (e.g. Gisser and Sanches, 1980; Provencher and Burt, 1994).

Finally, in the policy module the most desirable outcome is determined, based on the economic efficiency criterion and criteria such as social equity and ecological sustainability. Besides the suitability of policy instruments to achieve the desired outcome is reviewed against criteria such as effectiveness, acceptability and the transaction costs involved.

[1] Conventional cost-benefit analysis has been criticised on the ground that efficiency is not the only criterion for water policy review: social aspects might also be important in this respect. On the basis of *multiple objective analysis*, policies can be investigated with respect to a broader range of criteria, such as equity. Scores on criteria can be weighted according to social preferences. This method has been applied for an integral evaluation of alternative water management scenarios in the east of Gelderland (Ancot and van de Nes, 1981).

1.6 Outline of the thesis

Given the problem of desiccation of nature in the Netherlands, this thesis focuses on groundwater level and extraction management and their effects on agriculture and nature. Table 1.1 gives an overview of the subjects studied in the various chapters, and the kinds of analyses and micro-economic analysis techniques used.

Table 1.1 Overview study objects, and the kinds of analyses and techniques used in the various chapters.

Chapter	Object of study	Kind of analysis	Technique
2	Causes of desiccation of nature	Conceptual	Descriptive analysis
3	Uniform groundwater level management	Static modelling	Cost-benefit analysis
4	Non-uniform groundwater level management	Static modelling	Cost-effectiveness
5	Groundwater extraction management	Dynamic modelling	Optimal control theory
6	Instruments for groundwater management	Review assessment	Analytical analysis

In Chapter 2 the economic and institutional aspects of groundwater management will be discussed in order to gain an insight into the fundamental economic and institutional causes of desiccation of nature in the Netherlands.

In Chapter 3 uniform groundwater level management in agricultural areas with special ecological value is studied. The term uniform is used, because the same groundwater level applies to the whole area (see Figure 1.6). In the Netherlands agriculture is currently not able to maintain the various functions of farmland and of groundwater in a non-conflicting manner. The conflict between the productive and the ecological functions of farmland and groundwater reflects a conflict between private and public interests. Although most farmland is privately owned, the environmental amenities exhibit public good characteristics. The question is therefore: what groundwater level is optimal in terms of economic efficiency for society? Chapter 3 answers this question by analysing the trade-off between the value of ecological and of agricultural functions. Cost-benefit analysis is used to study socially optimal groundwater levels in agricultural areas by ascribing given monetary values to special ecological benefits provided by agricultural nature management. The trade-off between the agricultural production value and the monetary value of ecological benefits as a result of a shift from privately to socially optimal groundwater levels is studied. The analysis is applied to a study area in the eastern part of the Netherlands. The role of charges and compensation payments in achieving socially optimal groundwater levels in an equitable manner is studied. At this optimum, the groundwater level is raised to the point where the marginal costs of sacrificed production losses to agriculture equal the marginal benefits of environmental improvements. A comparative static analysis is justified because the average annual values of services are compared.

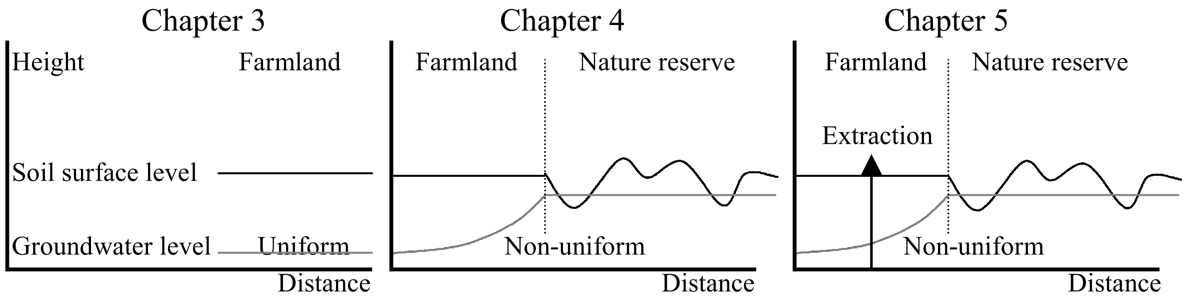

Figure 1.6 Groundwater management is studied in three different settings in the Chapters 3, 4 and 5.

In Chapter 4 non-uniform groundwater level management in nature reserves and adjacent farmland is studied. The term non-uniform is used, because groundwater levels in nature reserves and in farmland are usually different (see Figure 1.6). A higher groundwater level is desirable in nature reserves than in farmland. The groundwater level in nature reserves is strongly dependent on water management in adjacent farmland, which may have declined due to intensified drainage. Subsidies for restoration projects aimed at reducing the desiccation of nature are, at present, usually not allocated in the most effective way[2], due to a lack of insight into the cost-effectiveness of restoration. This information is, however, helpful to evaluate requests for project subsidies. Chapter 4 provides insight into the cost-effectiveness of restoration of nature and the division of payments among projects that reduce the desiccation of nature in such a way that restoration is maximised. Cost-effectiveness analysis instead of cost-benefit analysis is used, because the latter requires the quantification of benefits to nature in monetary terms, whereas reliable monetary estimates of such benefits were not available for the specific case study areas and benefit transfer is questionable. Different objective functions have been formulated, which maximise (1) the total increase in the unaffected surface area; (2) the total increase in the ecological conservation value; and (3) the total utility of an increase in unaffected surface area. Cost-effectiveness analysis is also used to compare investment costs of hydrological measures to re-wet nature. For the analysis, a multidisciplinary decision support model is developed, in which hydrology, agronomy, ecology and economy are integrated. The model is applied to evaluate subsidy requests of three characteristic reserves in the Netherlands. The model is useful to analyse the economic and ecological implications of restoration. A comparative static analysis is justified because the average annual values of services are compared.

[2] From 1995 until 2000 the allocation of subsidies among restoration projects under the former subsidy regulation (the GEBEVE regulation) was not based on a comparison of the cost-effectiveness of projects. The number of subsidy requests fell short of expectation and financial resources were sufficient to remunerate all requests that met some general prerequisites (see Article 7 of the Regulation of 14 November 1994, *Staatscourant* 223, Nr. J. 9416297). In the future the allocation of subsidies for restoration projects will come under a new interdepartmental subsidy arrangement for area-oriented policies (the so-called SGB regulation).

In Chapter 5 groundwater extraction in agricultural areas is studied. The impact of groundwater extraction on groundwater levels and groundwater quality is analysed (see Figure 1.6). In the Netherlands, the interdependency between water quality and quantity is an interesting issue to study because, although there is sufficient water available, often water of adequate quality is not found at the appropriate place and time. The deterioration of water quality reduces its usability, at the same time changes in quantity may affect its quality. Chapter 5 deals with the impact of agricultural shallow groundwater extraction on nature and groundwater quality by means of optimal control theory. Since groundwater extraction may affect the availability of groundwater of a certain quality for future generations, a dynamic approach is required to study extraction paths. Optimal control theory is used to study optimal agricultural groundwater extraction paths considering desiccation of nature and contamination due to extraction. The sum of discounted agricultural benefits and environmental damage is maximised over an infinite time horizon, taking into account the shadow prices of changes in the quantity and quality of the groundwater stock over time. In contrast to other approaches, changes in the quantity and quality of the groundwater stock are assessed simultaneously, because these factors are interactive. The optimal groundwater extraction tax that will induce farmers to behave in the socially optimal way is studied. The maximum principle technique is used to solve the continuous-time optimal control problem.

In Chapter 6 the suitability of current and alternative policy instruments for groundwater level management and groundwater extraction management will be reviewed against a number of performance criteria. This is interesting since current policy instruments for agricultural groundwater management do not seem to be efficient, while there is a lack of insight into the potential role of alternative policy instruments for groundwater level and extraction management. With increasing conflicting interests with respect to groundwater management across sectors, the need for efficient, equitable and sustainable groundwater management policies has increased in importance. As policy reform is conditional upon the size of the efficiency gains relative to the transaction costs involved, special attention will be paid in an analytical way to the transaction costs of policy reform.

Chapter 7 contains the summary and conclusions. The four research questions will be answered by summarising the main results of the study. Finally, policy recommendations and recommendations for future research are given.

2 Economic and institutional aspects of groundwater management

2.1 Introduction

Insight into the economic and institutional aspects of groundwater management is helpful in understanding the causes of the desiccation problem in the Netherlands and for making the right choices about groundwater management. To characterise groundwater problems in the Netherlands, first a general description of four kinds of groundwater problems is given.

1) Conflicting interests with respect to the management of in situ services of groundwater
In addition to extractive services, groundwater also provides *in situ* services, which occur as a consequence of groundwater remaining in place. For instance, the capacity of groundwater to prevent subsidence of the land, buffer against periodic water shortages, protect against seawater intrusion, protect water quality by maintaining the capacity to dilute, and to facilitate habitat and ecological diversity. Groundwater fulfils various functions in society (de Groot, 1992) – e.g. for agriculture, nature, navigation, fishery and recreation – which might conflict with respect to the management of *in situ* services characterised by e.g. groundwater levels. To study the optimal management of *in situ* services, a trade-off between the values of these functions has to be made. Economic literature on the optimal management of *in situ* services of groundwater is limited, since they are hard to value.

2) Externalities imposed by extractive services of groundwater
Extractive services of groundwater fulfil the consumptive needs of waterworks, households, industry and agriculture, such as production and sprinkling. Groundwater extraction may cause all kinds of externalities, such as depletion, the desiccation of nature and the subsidence and degradation of soils in irrigated areas e.g. due to waterlogging and salinisation. Externalities are unintended side effects of one party's action on another party that are ignored in decisions made by the party causing the effects. Demand management can tackle such problems. Externalities could, for instance, be internalised in the costs of groundwater. Economic literature with respect to these kinds of problems is widely available. Some studies focus on improving water use efficiency by adopting water-saving irrigation technologies (e.g. Caswell and Zilberman, 1985; Zilberman et al., 1994; Shah et al., 1995a; Burness and Brill, 2001). Other studies focus on optimal extraction paths (see Chapter 5). Waterlogging and salinisation problems have also been extensively studied (e.g. Shah and Zilberman, 1991; Shah et al., 1995b; Lee and Howitt, 1996; Wichelns, 1999).

3) Groundwater scarcity problems

Scarcity is a relative concept. In most countries the imbalance between water demand and supply can be bridged through management reforms. Challenges posed by growing scarcity can be addressed through two strategies: *supply management*, which involves developing new supplies, and *demand management*, which promotes policies that make better use of existing supplies (Winpenny, 1994). The appropriate mix of supply and demand management will vary with the levels of development and water scarcity. Randall (1981) introduced the term 'maturing water economies', which includes two phases. The *expansionary phase* is characterised by *supply-side* interventions. The *mature phase* is characterised by absolute water scarcity and a shift towards *demand management*, which treats water as an economic good.

4) Groundwater quality problems

Many problems arise because groundwater is not of the desired quality Water quality and pollution prevention have received a lot of attention. In particular the reduction in the use of pesticides and chemical fertilisers and in the disposal of animal manure reduces agriculture's contribution to water pollution.

This thesis focuses on the optimal management of *in situ* services related to *groundwater levels* and on *agricultural groundwater extraction,* because the desiccation of nature in the Netherlands is mainly the result of increased drainage and groundwater extraction. The desiccation problem in the Netherlands is different from most international water problems, which mainly concern scarcity and quality issues (see the third and fourth problem). It is, however, not a unique problem. In Hungary, for example, between the Danube and Tisza rivers, agricultural activity and droughts, particularly in the early 1990s, have led to declining shallow groundwater levels, threatening wetlands (OECD, 2000).

Public intervention in the water sector is often inevitable due to market failure, which is the divergence between the market outcome (without intervention) and the economically efficient solution. It may be the result of externalities and the public good nature of groundwater services. Public intervention is also often inevitable due to the common-pool nature of the resource, i.e. in case there are no well-defined property rights. It is important to note in this respect that a well-managed water sector needs a balance between private and public involvement, recognising the limits of the market and government, and the characteristics of water as a partly private and partly public good.

Imposing taxes on water is a politically sensitive issue for several reasons. Access to clean water is often seen as a basic right of all human beings (ICWE, 1992), because it is often considered as too vital to humans to be left to the economic forces of profit-maximisation

(Gibbons, 1986). Besides, many societies believe that water has special cultural, religious and social values (FAO, 1994). Goals other than economic efficiency, like social equity and ecological sustainability, are often guiding principles. This explains why the governments sometimes subsidises those uses of water that have a high value, but low ability to pay (Hartwick and Olewiler, 1998). It is therefore a challenge to identify the right balance between water treated as an economic good and water considered as a social good.

2.2 Valuation concepts

The full value of groundwater is the sum of use values (economic values) and non-use values (intrinsic values) (Figure 2.1). Use value is the value of the actual or potential use of a good. Non-use value is the value that is attached to neither direct nor potential use of the good.

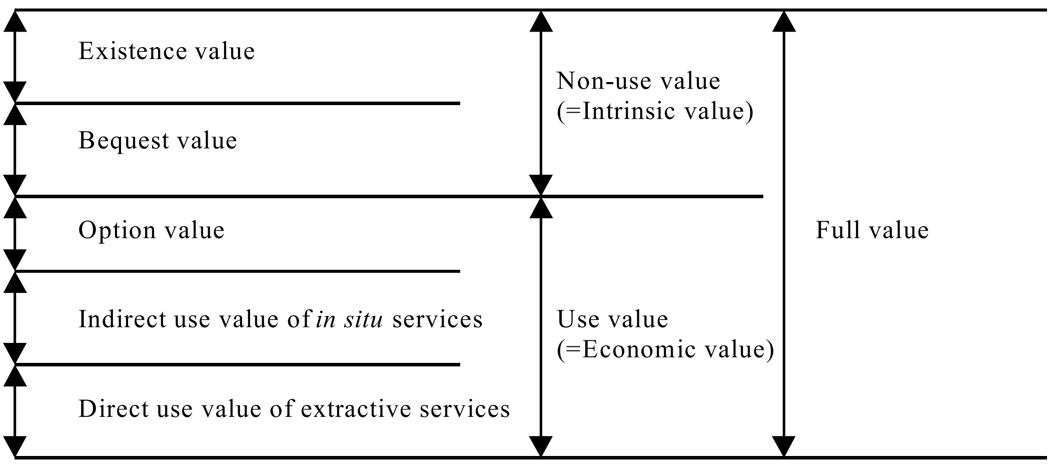

Figure 2.1 Concept to determine the full value of groundwater (after: GWP, 2000).

Use values consist of direct use values, indirect use values and option values (CVG, 1997). Direct use means consuming groundwater. Extractive services imply off-stream rival use: they often only provide marketable private goods, but also may affect *in situ* services. Indirect use means using rather than consuming water. *In situ* services imply in-stream non-rival use: they provide non-marketable public goods as well as private goods. Option value represents a willingness to pay now to preserve an option for gaining a benefit in the future.

Non-use values are independent of the present use of the resource and consist of bequest and existence values. Bequest value may represent a value of preserving a resource for future generations. It represents the value that an individual assigns for the benefit that others will gain from future use. Existence value is the value someone attaches to knowing something exists, independently of any value associated with actual or potential use.

This groundwater valuation concept provides a basis for evaluating trade-offs between various values of *in situ* services related to groundwater level management. The marginal costs of changes in groundwater depth can be used in trade-off analysis to determine optimal situations, as shown in Figure 2.2. The horizontal axis shows the groundwater depth, which can be attuned to agriculture for production purposes or to nature for ecological services. Within certain ranges, shallow depths are desirable for nature, whereas deeper depths are desirable for agriculture. At 0.2 m bss, the groundwater depth is fully attuned to nature and the marginal costs to nature MC_r are zero in our example, whereas at 0.8 m bss, the groundwater depth is fully attuned to agriculture and the marginal costs to agriculture MC_a are zero. The line sloping upward to the right depicts the marginal costs to nature of attuning groundwater depth more to agriculture. At point D^*, the marginal costs to agriculture and nature are equal, which is the optimal groundwater depth in terms of economic efficiency. At point D^1, the groundwater depth should be more attuned to agriculture, since the decrease in marginal costs to agriculture exceeds the increase in marginal costs to nature. At point D^2, the groundwater depth should be more attuned to nature, since the decrease in marginal costs to nature exceeds the increase in marginal costs to agriculture.

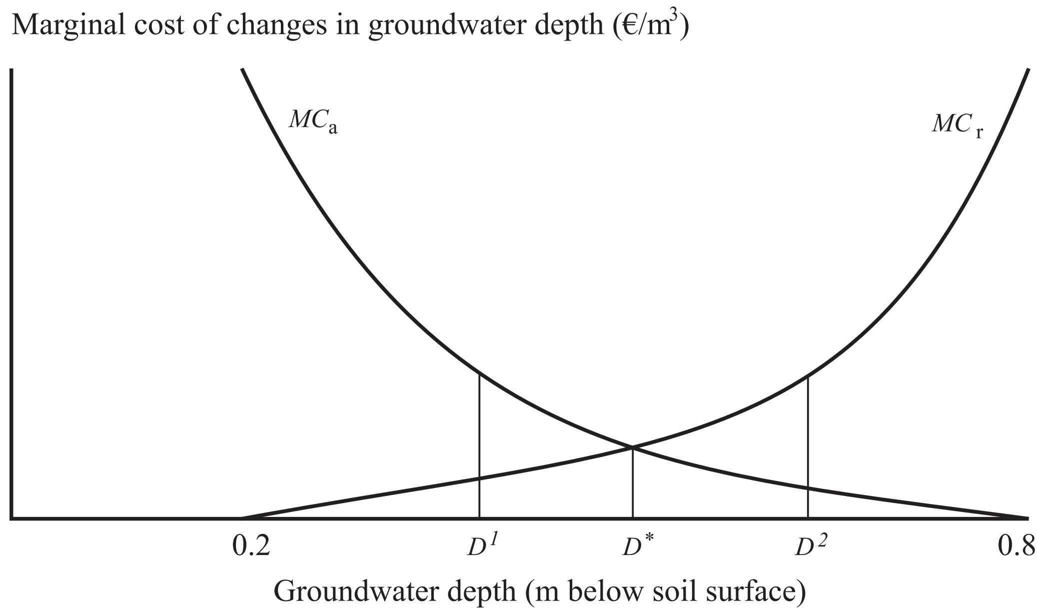

Figure 2.2 Marginal costs to agriculture MC_a and nature MC_r as a function of groundwater depth.

The value of extractive services of groundwater can be derived from the values of marketed products and tends to accrue to individual users, whereas the value of the *in situ* services of groundwater is often hard to determine, since such services are not marketed. Farmers receive no signal through the market to take the value of losses of nature into account in their decision-making. This explains why *in situ* services tend to be underpriced and mismanaged.

Monetary valuation

To analyse socially optimal groundwater management, a trade-off between the agricultural production value and the value of nature has to be made. Two steps are needed to determine these values. Firstly, the impact of changes in groundwater management on crop growth and nature development has to be determined. Secondly, the monetary values of agricultural products and nature have to be determined. Because agricultural products are private marketed goods, their monetary value can be derived relatively easily. The monetary valuation of nature, however, is not easy to establish, because many of the goods and services it provides are not marketed. An additional complication is the fact that the value ascribed to nature changes with income levels and with the total size of nature reserves in a country. It is especially important to consider the monetary values of nature if a high value is ascribed to nature. This is currently the case in the Netherlands, since almost all-political parties consider nature conservation and restoration desirable. This study, however, does not aim to estimate the monetary values of public benefits. In this thesis benefit transfer is used to determine the monetary values of public benefits. A given monetary value of the public benefits of agricultural nature management is used, which is estimated by Brouwer and Slangen (1998) by means of the contingent valuation method.

Various methods can be used to estimate the monetary value of ecological benefits. The economic background, and the applicability and appropriateness of these methods are discussed in van Ierland and de Man (1993).

- The contingent valuation method, which is based on the willingness to pay in hypothetical situations, seems to be the most appropriate method to estimate the monetary value of benefits from nature management, since it is able to measure both their use and their non-use value, contrary to other valuation techniques.

- The travel cost method and hedonic pricing method have the advantage of being based on actual behaviour, but neither method can estimate non-use values (cf. Pearce and Moran, 1994; Perman et al., 1999). Especially non-use values are expected to make up an important part of the value of the amenities found on farmland, i.e. rare plant species (Brouwer and Slangen, 1998).

- Another valuation technique – the production function approach – generally uses scientific knowledge on cause-effect or dose-response relationships, i.e. the relationship between environmental quality variables and the output level of a marketed commodity. The production function approach is unsuitable for estimating the benefits obtained from nature management, unless there is an associated market good or service with which to link dose-response functions (Spash and Carter, 2001).

2.3 Economic aspects of groundwater level management

The desiccation of nature caused by intensified drainage is the result of *conflicting interests* with respect to the management of *in situ* services, rather than *competition* for extractive services. It reflects a conflict between the private and the public interests served simultaneously by groundwater. The problem can be ascribed to ignoring the values of particular *in situ* services of groundwater, when making private decisions about groundwater level management. To study optimal groundwater level management, a trade-off between the values of all *in situ* services has to be made. It is important to note that it is hard to internalise a reduction in the value of services that serve public interests in the price of *in situ* services that serve private interests, because *in situ* services often have no price. Users do not pay for usage as such, but for the management of *in situ* services, since they are just used and not bought in the market.

Groundwater can be divided into four categories on the basis of rivalry and excludability (see Table 2.1). Groundwater usage is non-rival if it does not exclude others from usage and is non-excludable if it is not possible to exclude users.

Table 2.1 Groundwater can be divided into four categories. Groundwater is non-rival if its consumption involves zero marginal cost, and non-excludable if agents cannot be excluded from it.

	Excludability	
Rivalry	Excludable	Non-excludable
Rival	Private goods	Common resources
Non-rival	Club goods	Public goods

Groundwater used for sprinkling is a *private good*, whereas in the absence of institutional constraints, groundwater is a *common resource* (see Section 2.4). Many of the *in situ* services of groundwater can be considered as *local public goods*, since they are non-rival and non-excludable. Groundwater is a *club good* when beneficiaries can be excluded from non-rival usage at low costs, for instance by an access fee (Musgrave and Musgrave, 1976). Private firms can supply club goods, while the public provision of public goods is usually desirable to avoid under-provision and since it is hard to recover the costs of public goods from users; there will be free riders.

2.4 Economic aspects of agricultural groundwater extraction management

The desiccation of nature caused by increased agricultural groundwater extraction can be characterised as an externality problem, since extraction affects the *in situ* service in such a way that it imposes a cost on nature. This externality is usually not considered when making decisions about agricultural groundwater extraction, because water's value for nature is often not taken into account. Externalities violate the conditions for optimal allocation of resources in the economy. *Underpricing* of groundwater often occurs in the presence of externalities. Usually only supply costs are reflected in the price and consequently groundwater tends to be abused. The externalities, although noticed, are left unpriced and hence the bearers are, normally, uncompensated in the private market environment. If externalities are priced and bearers are compensated, then externalities are internalised.

In the absence of institutional constraints, groundwater is a *common property resource* (Kneese, 1995). As the 'tragedy of the commons' highlights, no single user has an incentive to refrain from exploiting it, because others would continue to do so (Winpenny, 1994). Resources are, however, to some extent characterised by exclusiveness, because only those who are situated above the aquifer are able to extract from it. Stevenson (1991) provides an interesting discussion about the successful exploitation of common resources. User rights can prevent over-exploitation. Often, however, rights are not properly defined, which may limit the possibility of reallocation. Coase (1960) stresses the importance of the existence of well-defined water rights. He reduces market perfection conditions to two issues and shows that market allocation will be efficient given low transaction costs and given well-defined water rights, regardless of the allocation of rights. Well-defined water rights are, however, hard to establish, since groundwater is *not a homogeneous* product. It is characterised by a bundle of attributes – such as location, quality and timing of supply – which should be studied in interdependency of each other. The location determines its accessibility, the quality determines its usability and the timing governs its reliability (FAO, 1994).

The consequences of not considering the negative externalities of agricultural groundwater extraction in the price of groundwater when making extraction decisions is shown in Figure 2.3. If there are negative externalities, the marginal social costs of agricultural groundwater extraction will be higher than the marginal private costs. Farmers will use water as long as the private benefits of an additional unit exceed private costs. This means that farmers will demand the quantity of groundwater E^p at a price v^p. From a social point of view, farmers extract too much in the presence of negative externalities and water is underpriced. External costs that have to be borne by society in the private optimum E^p are shown by the triangle ABC. If externalities are internalised in the price of water, farmers will face a price v^* and extraction will be reduced to the socially optimal extraction level E^*, where marginal benefits equal marginal social costs. Figure 2.3. shows that the decrease in costs (the area DBE^pE^* under the marginal social cost curve) exceeds the decrease in benefits (the area DCE^PE^* under the marginal benefit curve) and there are social welfare gains (shown by the triangle DBC).

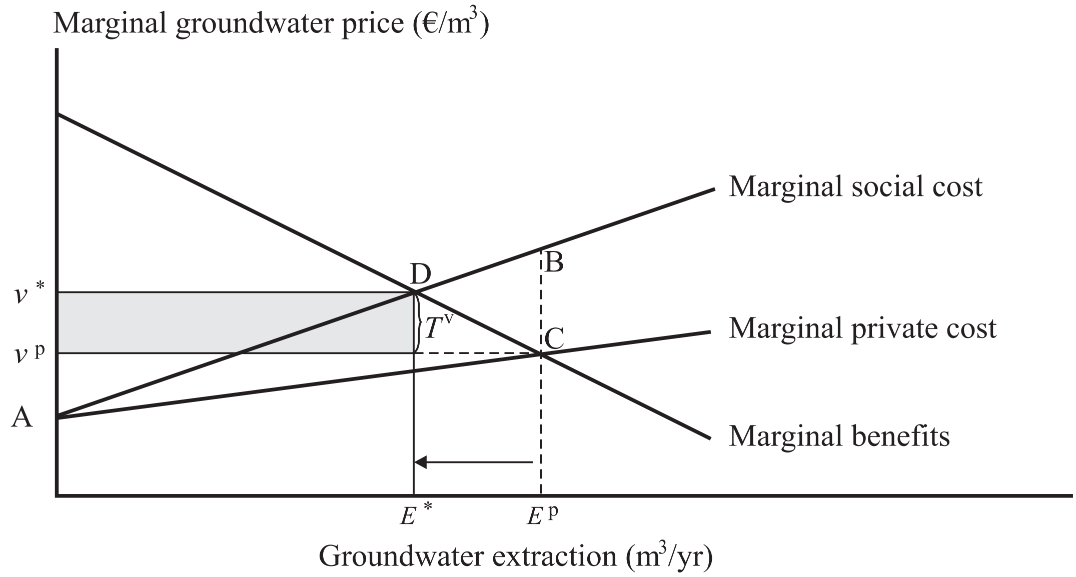

Figure 2.3 Social welfare gains of the internalisation of negative externalities in the price of groundwater.

The value and cost of water should not be confused with the price of water, which is the amount of money actually paid for it. The price does often not reflect the value and cost of water. Often there is a ‘rent’, which is the difference between the actual price paid for water and the value of water, i.e. the maximum amount the user would be willing to pay for usage. In practice, the actual price paid is often lower than the value, because social and political goals often override economic criteria.

There are some general principles involved in assessing the costs associated with the provision of water; these are shown in Figure 2.4. The full cost of providing water includes the full use cost and non-use cost.

The full use costs consist of the full supply costs, the opportunity costs of alternative water use and costs of externalities. The full supply costs include the operating and maintenance cost and capital charges. The opportunity cost is the value forgone, when groundwater is used for one purpose instead of for its next best alternative use. Two types of externalities can be distinguished: economic externalities – which arise from changes in the economic activities of affected sectors – and environmental externalities associated with public health and ecosystem maintenance. It is usually more difficult to assess environmental externalities than economic externalities, especially in situations involving conflicts (GWP, 2000). The perpetrators of externalities usually evaluate damage less severely than other interest groups.

The non-use costs are independent of the present use of the resource. It concerns costs that arise independent of water use.

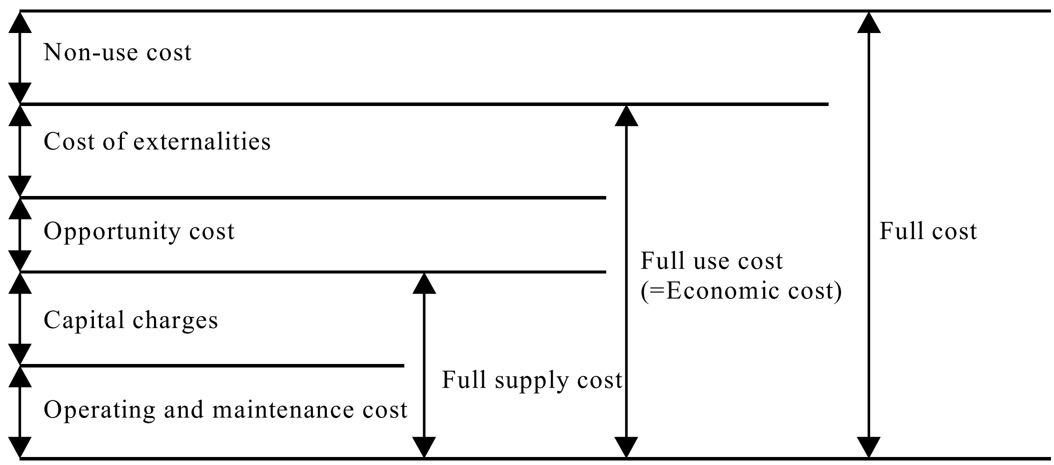

Figure 2.4 Concept to determine the full cost of the provision of water (after: Rogers et al., 1998).

Briscoe (1996) showed that the size of the various cost components varies quite widely among sectors. The full supply costs of urban water are for instance relatively high, while opportunity costs are low. This is in contrast to the situation for agriculture, which faces modest full supply costs, but high opportunity costs if there is competition from urban use.

There are large differences among countries in the way the price of water covers costs. In some countries, the price does not even cover operation and maintenance costs. In other countries capital costs are included as well, and sometimes an effort is made to include opportunity cost and cost of externalities in the price of water (Dosi and Easter, 2000).

Agricultural groundwater extraction in the Netherlands

In the Netherlands agriculture accounts for about 10% of all groundwater extracted, but locally this percentage is much higher. Total groundwater extraction by waterworks, industry and agriculture is according to Feddes et al. (1999) about 900, 200 and 100 million m^3/yr, respectively. According to Dufour (2000) about 470 million m^3/yr is used for domestic consumption and 300 million m^3 of the water extracted each year by waterworks is used for industrial purposes, while industries extract 340 million m^3/yr from their own wells.

Table 2.2 shows that agricultural groundwater extraction varies from 173 million m^3/yr in dry years to 40 million m^3/yr in wet years. It covers about 60-80% of agricultural water use. Although individual extractions for sprinkling are relatively small, the great number of farmers applying sprinkling makes total water demand rather large. Especially agricultural extractions contribute to the desiccation problem, since these are shallow extractions (2 to 3 m bss). In 1997 about 73% of agricultural groundwater extraction was sprinkled on grass

(Meeusen et al., 2000). The amount of water used for sprinkling depends not only on the amount of precipitation in the growing season, but also on the management practice of farmers (such as risk attitude) and on how stringent extraction policies are.

Table 2.2 Water usage by agriculture for sprinkling in the Netherlands (excluding horticulture; million m^3/yr).

	1994/1995	1995/1996	1996/1997	1997/1998	1998/1999	1999/2000
Groundwater	125	173	140	72	40	54
Surface water	32	50	67	11	9	15
Other sources	18	37	23	7	4	7
Total	175	260	230	90	53	76

Source: LEI-Dutch Farm Accountancy Data Network, various years.

The marginal value of sprinkling varies, due to the site-specific characteristics of production (such as climate, soils, crops and technology used). Values may also differ according to the framework employed to make the estimate (short term versus long term; public versus private accounting). In the Netherlands the sprinkling of grassland is often seen as a low-value application which in some cases is even not remunerative.

A review by Dijk et al. (1994) of the literature on the profitability of sprinkling in the Netherlands shows that studies produce different results. According to farmers, the sprinkling of grassland is not a low-value application, because benefits other than changes in output should also be considered in the calculations. Especially on drought-sensitive soils (e.g. sandy solids) sprinkling may be essential for good farm management to avoid a long restoration period of grass due to drought damage to the roots. Besides, sprinkling leads to the better utilisation of minerals, which is not only beneficial for crop growth but also reduces leaching problems. These benefits are generally not considered in the profitability calculations, which therefore underestimate the benefits of sprinkling.

2.5 Institutional aspects

In this study groundwater will be treated as an economic good. This, however, does not automatically mean that economic instruments, such as taxes, should be used in practice to achieve optimal situations, as is often believed (Savenije, 2000). Other instruments can also be very cost-effective and suitable. Various policy instruments for groundwater management can be used by the government to achieve policy targets, i.e. to reduce the desiccated area. They are classified here in three categories:

1) *The institutional and legal environment* in which water is supplied and used can be changed. Institutions set the 'rules of the game' within which the economic system

operates and help define the rights, privileges and responsibilities that guide human activities. The water rights system is a water institution since it includes provisions which determine access to water (Bromley, 1991). The role of water management institutions for wetland restoration has been studied by Adger and Luttrell (2000) and Hodge and McNally (2000). With the term institutional change we refer in this thesis not only to changes in the water rights system, but also to changes in the institutional system that balance the interests of stakeholders with respect to water management.

2) *Environmental policy instruments for water management* are usually divided into market-based and non-market-based devices (see Winpenny, 1994; Rosegrant, 1997). Since non-market-based devices take a variety of forms, a distinction is made here between regulatory and persuasive instruments. The following division into three classes – based on the way in which a government can influence an actor's behaviour – is often made (see Turner and Opschoor, 1994).

- *Economic instruments* are market-based. They influence indirectly the behaviour by providing incentives to use water efficiently. Four types can be distinguished:
  - Taxes or charges: a straightforward way to put prices on the use of the environment.
  - Tradable rights or marketable permits: environmental quotas or ceilings on extraction that, once initially allocated by authorities, can be traded subject to prescribed rules.
  - Deposit-refund systems: a deposit is paid on potentially contaminating products.
  - Subsidies (usually in the form of grants): in general, these are incompatible with the Polluter-Pays Principle and can only be justified as a transitional measure.

- *Regulatory instruments* (also known as command-and-control instruments) can directly restrict, forbid and/or prescribe action. The main feature is that there is no legally free choice for actors, who face penalties. Examples are permits, standards and quotas.

- *Persuasive instruments* work even more indirectly than economic instruments, because actors are supposed to take action of their own accord. There are no sanctions. Examples are education, extension, information, negotiation and voluntary agreements.

3) *Direct interventions,* such as restoration measures, work in various ways. Measures can prevent the cause (e.g. through less drainage or groundwater extraction), counteract the effect (e.g. through the inlet of foreign surface water), or separate the functions (e.g. by creating hydrological buffer zones or setting up partitions of foil in the soil). The suitability of measures depends on the kind of ecosystem and the cause of the problem (kind of hydrological change).

Policy instruments for demand management that have been widely studied under different socio-economic and physical environments will be discussed here, namely water pricing, tradable water rights and quantity-based controls (Strosser, 1997; Rosegrant, 1997). These instruments can reduce extraction to the social optimal level. See Bhatia et al. (1994), FAO (1994) and Winpenny (1994) for a discussion of a larger number of instruments.

Although from a theoretical point of view *water pricing* is the basic instrument that helps to distribute limited resources to users in an efficient way, it does not always function in that way. Evidence that agricultural demand for water is sensitive to water prices is sparse, but this might be due to very low water prices in many countries (Winpenny, 1994). Agricultural water demand is usually inelastic up to a certain price level (Garrido, 1999). This price threshold depends on the productivity of water, the set of production strategies, the proportion of land devoted to permanently irrigated crops, and the irrigation technologies. Prices should be high enough to move into the elastic part of the demand curve. A *tax* can be levied on each unit of groundwater extracted in order to internalise extraction externalities. Figure 2.3 shows that at the price v^p, a tax level T^v can reduce extraction to the socially optimal level E^*. The shaded rectangle shows the tax revenue.

Tradable water rights can also reduce groundwater extraction to the socially optimal level. This market approach is appealing since it combines an advantage of the quantity-based system with the cost advantage of a tax system (Pindyck and Rubinfeld, 1989). The agency that administers the system determines the total number of rights and therefore the total amount of extraction, just as quantity-based systems would do. But the marketability of the rights allows extraction reduction to be achieved at minimum cost, just as a system of taxes would do. The objective will still be reached even if new parties enter the market, since the total number of rights is limited. Also, it offers nature organisations the opportunity to buy rights in order to reduce extraction (Pearce and Turner, 1990). According to Strosser (1997) requirements for well-functioning water markets include water scarcity, well-defined and transferable water rights, a large number of purchasers and sellers, no or limited transaction costs and the existence of an appropriate information system. Since these requirements are often not met and as a result of potential third-party effects of transaction that may arise, water markets have failed to develop in many areas or function at the margin with a low number of transactions and small volumes transacted.

Quantity-based control mechanisms, like extraction quotas, specify a legal limit on how much water may be extracted. Extraction quotas are usually assigned in proportion to extraction in a base period or based on the proportion of land overlying the aquifer. Quotas do not provide incentives to use water more efficiently, but constrain extraction to the socially optimal level (under full information), by imposing substantial monetary penalties for exceeding the limits. If the price elasticity of water demand is equal or close to zero (inelastic), quotas will be more effective in constraining water extraction than water pricing. If the price elasticity of water demand is significantly different from zero and negative (elastic), quotas and water pricing lead to the same extraction reduction when the demand is constant. However, if increases in demand for water are expected, then a quota is preferred as it constrains extraction at the same level, while extraction will change under water pricing.

Economic instruments have a number of advantages (OECD, 1991). Firstly, they are static efficient, which means that extractions occur where it is most efficient to do so. Secondly, they offer an ongoing incentive to reduce extractions below the level that environmental policy prescribes. The property of economic instruments to offer more incentives to innovate than direct regulation, is called dynamic efficiency (Dijkstra, 1998). Thirdly, they increase flexibility. It is generally easier and faster to modify and adjust a tax than to change legislation or a regulation. Finally, they may generate revenues for the government.

In spite of these advantages, economic instruments are not widely applied in environmental policy for a number of reasons (cf. Dijkstra, 1998). There might be market imperfections, other than not considering environmental damage. Also, there might be an uncertain relationship between taxes and extraction. For instance, it is often believed that the price elasticity of demand is too low for water pricing to effectively restrain demand (Kneese, 1995). Moreover, economic instruments are less suitable when emergency problems necessitate a quick reduction in extraction or the government wants to ban a certain extractor. Further, economic instruments may not be widely applied because they are new or are politically sensitive for various reasons discussed in Section 2.1. Finally, transaction costs may be high relative to the size of efficiency gains.

The use of voluntary agreements also has a number of advantages. Firstly, they offer greater flexibility and freedom to find efficient solutions that are tailored to specific conditions, since they use the participants' specialised knowledge of local conditions. Such agreements take differences in the vulnerability of areas into account, which is essential when impacts on the environment are diffuse due to complex processes in the soil. Secondly, voluntary agreements encourage proactive cooperation, since they are the result of negotiations between groups who use the same resource but with divergent intentions. They can induce participation by providing positive incentives – like subsidies (the carrot approach) – or by threatening a harsher outcome, like bans (the stick approach). Finally, they have the ability to meet targets more quickly due to decreased implementation lags and reduce the transaction costs of the governmental bureaucracy (Kuks, 1998). The effectiveness depends on the magnitude of the background threat and allocation of bargaining power (Segerson and Miceli, 1998). It is typically a bottom-up approach (Heinz, 2001).

Policy review criteria
The economic efficiency criterion[3] is one of the basic principles recommended for policy review; the other criteria are effectiveness, administrative feasibility, equity and acceptability (OECD, 1991). However, a sixth criterion can be added to this list, namely technical efficiency, since some policies provide incentives for the adoption of modern technologies.

Economic efficiency (often referred to as allocative efficiency) is achieved when the net marginal values per unit of water are equal for all uses. This proposition is known as the equimarginal (Young and Haveman, 1985). Water transactions are interesting if differences exist between the net marginal benefits of water. There are two types of allocative efficiency. Inter-sectoral allocative efficiency can be achieved by diverting water away from one sector to another sector with a higher return to water. This is politically risky. Intra-sectoral allocative efficiency can be achieved by re-allocating water within a sector, and is politically less risky. There are two concepts to evaluate allocative efficiency (Perman et al., 1999):

- An allocation is *Pareto efficient* if there is no feasible allocation where everyone is at least as well off and at least one agent is strictly better off. In other words, there is no way to make one group of people better off without making another group worse off. Pareto efficiency provides no insight into the distribution of gains from trade.
- According to the *Neo-Paretian criterion* (Kaldor and Hicks' compensation principle), there are allocation profits if winners are able to compensate losers (benefits exceed costs). Real compensation does not have to take place. The *Neo-Paretian criterion* is less strong than the Pareto criterion. The former is the basis of cost-benefit analysis.

Environmental effectiveness indicates to what extent policies achieve their objectives. *Administrative feasibility* reflects how easy it is to implement, monitor and enforce policies. The associated costs are defined here as *transaction costs* of policy reforms.
Equity shows whether costs and benefits are equitably distributed among affected parties, which affects the *acceptability*. A policy is also more likely to be acceptable - will encounter less resistance - if it is aimed at tackling a severe problem, if there is a strong lead from prominent political figures, if it is accompanied by publicity and if people are well informed.
Technical efficiency is achieved if there is no way to produce more output with the same inputs or to produce the same output with less input. The technical efficiency can be improved by adopting modern water-saving irrigation technologies.

[3] The criterion of efficiency is attractive to economists because it carries little ethical content. There is, however, no reason to believe that resource allocations are ethically desirable just because they are efficient. Optimality can only be assessed in terms of social welfare, which is only meaningful if there are agreed ethical principles. An optimal arrangement is efficient, but an efficient arrangement is not necessarily an optimal one. Inefficient solutions can therefore be socially preferred. The terms private and social optimum used in this research are, however, not based on agreed ethical principles.

Responsibilities of the various governmental levels with respect to water management

The division of strategic and operational water management tasks among governmental levels in the Netherlands is shown in Table 2.3. The central government and provinces exercise general management, while the water boards exercise functional government.

Table 2.3 Distribution of tasks in water management among governmental levels in the Netherlands.

Governmental level	Central government		Provinces		Water boards	
Management task	Strategic	Operational	Strategic	Operational	Strategic	Operational
Surface water quantity	x	x	x	x		x
Surface water quality	x	x	x	x		x
Groundwater quantity	x		x	x		
Groundwater quality	x		x	x		

Source: Perdok and Wessel (1998).

Central government takes care of the waters of the national system and supervises the management of regional and local waters and groundwater resources. The majority of the water management tasks are the responsibility of the Ministry of Transport and Public Works (Verkeer & Waterstaat; V&W). This ministry is responsible for all water systems, and its policy is to coordinate all the claims of the different interests on the water system, i.e. to perform integrated water management. Two other ministries are involved. The Ministry of Housing, Spatial Planning and the Environment (Volkshuisvesting, Ruimtelijke Ordening en Milieu; VROM) is responsible for the protection of ecological values and for the drinking water supply. The Ministry of Agriculture, Nature Management and Fisheries (Landbouw, Natuurbeheer en Visserij; LNV) also embodies two conflicting interests. The LNV's Department of Nature Management protects ecological values, whereas its Department of Agriculture represents the interest of agriculture (Boogerd et al., 1997).

The *provinces* are responsible for the water policy in their territory. Most provinces have delegated the majority of the surface water quantity and quality management tasks to water boards. Groundwater management is entirely in the hands of the provinces. They cannot delegate the operational groundwater resource management. Provinces are empowered to establish and abolish water boards and to determine the territory and water management tasks of the water board, as well as the composition of the governing body and the way its members will be chosen. Provinces also supervise the work and finances of water boards. The 1983 *Groundwater Act* (implemented in 1985) prescribes that provinces have to draw up groundwater management plans and rules for issuing permits and levying charges on the amount of groundwater extracted (Blumenthal and Visée, 1988). Local differences in the availability and quality of the groundwater resource and the role groundwater plays with respect to the terrestrial ecology have created a diverse system of rules for charges. Tariffs and the charge-free threshold vary among provinces and are subject to change.

Since *water boards* are responsible for balancing the different – often conflicting – interests of water management, their tasks and the impact of changes in their composition on decisions about the water level will be discussed in more detail.

Water boards can have one or more of the following tasks: water control, water quantity management, water quality management and the management of inland waterways and roads. Dutch water boards are to a large extent self-supporting and have a total annual turnover of €2.3 billion. The integration of water management tasks will require water boards to become increasingly oriented towards general management rather than the mere functional management of a limited number of tasks on behalf of a limited number of interest groups. Water boards have to balance the different interests at stake with respect to water management in order to make decisions about surface water levels, which directly affect groundwater levels, whereas they are not responsible for groundwater level management. Water boards find it logical that they should become responsible for operational groundwater management, since they regard it as their responsibility to manage water in an integral manner (Union of Water boards, 1999).

Before the adoption of the *Water Authorities Act* in 1992, mainly agricultural interests dominated the governing body of water boards as a result of the strong financial dependence. Water quantity management consequently mainly served the interests of agriculture. The interests of nature were poorly represented until 1992 (Perdok and Wessel, 1998). This changed after the *Water Authorities Act* introduced two new interest groups: residents and tenants in addition to the already three existing interest groups, i.e. landowners (e.g. owners of farmland and nature reserves), property owners (e.g. home-owners) and the owners of company premises. All these groups may propose candidates for the governing body, since the basic principle of water boards is ‘interest, payment and authority’, indicating that only those groups who have an interest in the tasks of the water board and pay for it are represented in the governing body. Provinces determine the number of seats to be held by various interest groups. The elections serve to select individual members, not to determine the ratio between interest groups in the governing body (as in general elections). This means that candidates for a certain seat can only be chosen by the people represented by this seat. As environmental and nature organisations, which belong to the residents group, have a relatively strong lobby, they are now better represented within the governing body of water boards than they were before 1992. They represent the interests of nature. If more interests are represented within the water boards, various interests with respect to groundwater levels will be better balanced and can play an appropriate role in decisions on surface water levels. In various places, the water board is doing its best to repair the damage caused by declining water levels. A better-balanced water level in which water boards take into account interests of nature should prevent unnecessary damage (Union of Water boards, 1999).

3 Uniform groundwater level management in agricultural areas[4]

3.1 Introduction

In many agricultural areas with special ecological value, agriculture and nature have competing interests with respect to the management of the groundwater level. Especially since about 1950 the groundwater level in many agricultural areas in the Netherlands has been lowered, mainly by increased drainage and discharge of water to create favourable crop-cultivation conditions for agriculture. As a result, ecosystems susceptible to hydrological changes have been damaged, i.e. desiccated, which implies that species adapted to wet and moist environments have disappeared. This externality is not yet fully considered when determining optimal groundwater levels in agricultural areas with special ecological value, and consequently those ecosystems are affected.

Agriculture may play an important role with respect to nature management, since farmland often provides a habitat for plant species. There is, however, a lack of insight into the costs of agricultural production losses and the ecological benefits of agricultural nature management brought about by higher groundwater levels. The setting of the problem is the case of increased drainage of shallow groundwater.

Despite the importance of the desiccation problem in the Netherlands, the economic literature on modelling this externality has been limited, although much literature is available on modelling externalities from water management (e.g. Zilberman et al., 1993; Shah et al., 1995a, b). However, most of these studies are concerned with water scarcity problems and the allocation of groundwater among different users (e.g. Howitt and Lund, 1999) or with environmental problems caused by the disposal of agricultural drainage water (e.g. Shah and Zilberman, 1991). Only a few studies focus on the valuation of ecological benefits resulting from changes in agricultural water management (e.g. Loomis et al., 1991).

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[4] Chapter 3 is a modified version of Hellegers, P.J.G.J., K. Oltmer, E.C. van Ierland and L.C. van Staalduinen, 2001, An economic analysis of shallow groundwater management for nature conservation and agricultural production. *Journal of Environmental Planning and Management*, 44(4):545-559.

The main aim of this chapter is to develop a model to analyse the trade-off between agricultural production values and the monetary value of ecological benefits of agricultural nature management as a result of changes in the groundwater level. A comparative static cost-benefit analysis is performed to compare annual losses to agriculture and annual benefits to nature resulting from a shift from privately to socially optimal groundwater levels. The consideration of monetary values for both forgone agricultural production and ecological benefits introduces an inter-sectoral trade-off into the optimisation problem. Attention will be paid to the potential role of economic incentives as a tool to achieve socially optimal groundwater levels. The model will be applied to the eastern cattle-raising area in the Netherlands.

This chapter studies the economic module of Figure 1.5, i.e. how changes in groundwater depth affect agriculture and nature. Section 3.2 explains the optimisation model developed to analyse the inter-sectoral trade-off. Section 3.3 describes the data used in the case study. Section 3.4 presents the results of the empirical analysis in the study area, which serves as an illustration of our model. Insight is provided into the discrepancy between the privately and socially optimal situation. Section 3.5 contains the conclusions.

3.2 Optimisation model

In the Netherlands, both crop growth and nature development depend on the average groundwater level in the spring. The relationship is indirect, because crop growth and nature development are dependent on the availability of soil moisture, which in turn is determined by shallow groundwater levels through capillary rise. Such other factors as salinity, nutrient availability and acidity also influence crop growth and ecosystems. Higher groundwater levels restore nature, but may cause wet damage to agriculture.

Our model analyses the shift from the groundwater level that maximises the objective function of a private planner to the groundwater level that maximises the objective function of a social planner. The groundwater depth D relative to the soil surface is the decision variable. The optimal groundwater depth of the private planner is obtained by maximisation of the agricultural production value (equation (3.1)). The optimal groundwater depth of the social planner is obtained by maximisation of the sum of the agricultural production value and the monetary value of ecological benefits of agricultural nature management (equation (3.2)), which is based on a nature index N and an annual monetary value of nature V. Crop yields and the nature index both depend on the groundwater depth (equations (3.3) and (3.4)). The nature index measures the ecological benefits of agricultural nature management.

$$\max \sum_{i=1}^{i=m} Y_i P_i A_i \tag{3.1}$$

$$\max(\sum_{i=1}^{i=m} (Y_i P_i A_i) + NVA^{\mathrm{n}}) \tag{3.2}$$

Subject to:

$$Y_i = f_i(D) \qquad (f \text{ is concave}) \tag{3.3}$$

$$N = g(D) \qquad (g' < 0 \text{ and } g'' > 0\text{, i.e. } g \text{ is decreasing and convex}) \tag{3.4}$$

Where:

Y_i = Yield of crop i (tonnes/ha); for $i = 1,...,m$;
P_i = Price of crop i (€/tonne); for $i = 1,...,m$;
A_i = Area of crop i (ha); for $i = 1,...,m$;
i = Crop index;
N = Nature index (-);
V = Annual monetary value of nature for $N=1$ (€ ha^{-1} yr^{-1});
A^{n} = Agriculture area with special ecological value (ha) (with $A^{\mathrm{n}} \in \sum A_i$);
D = Groundwater depth relative to soil surface (metre bss).

Fixed agricultural costs (such as sowing costs) remain constant and need not be taken into account explicitly in the optimisation. Variable agriculture costs (such as harvesting costs) are affected by the groundwater depth, but changes in farm management practices will reduce these costs. In a more elaborated study these costs could easily be introduced into the analysis, but will hardly affect the results. Investment costs of hydrological measures to change the groundwater depth are not considered, since we assume that such adjustments in groundwater depths can be realised by means of existent infrastructure (at zero cost).

The following steps are taken to implement the model. The relationships between crop yields and groundwater depth (equation (3.3)) are estimated (step a) in order to calculate agriculturally optimal groundwater depths (equation (3.1)) (step b). The relationship between the nature index and groundwater depth (equation (3.4)) is determined and the monetary value of nature derived (step c) in order to calculate the socially optimal groundwater depth (equation (3.2)) (step d). The sensitivity of the results to the various values of three parameters is then studied (step e).

a) Relationships between crop yields and groundwater depth

The agronomic relationship between crop yields and groundwater depth is an inverted U shape, because there are yield losses if the groundwater depth is too shallow or too deep, with the optimum somewhere in between. In the first case, the availability of air and oxygen in the soil, the nitrogen supply, and the soil temperature are sub-optimal. In the second case, water supply might be insufficient to guarantee optimal crop growth (van der Schaaf and Witte, 1997). Equations (3.5) and (3.6) show, the quadratic and the cubic function; β_0, β_1, β_2 and β_3 are crop-specific parameters. The use of other inputs like fertiliser is assumed to be constant and their impact is reflected in the constant β_0 of the production function:

$$Y_i = \beta_0 + \beta_1 D + \beta_2 D^2 \tag{3.5}$$

$$Y_i = \beta_0 + \beta_1 D + \beta_2 D^2 + \beta_3 D^3 \tag{3.6}$$

b) Agriculturally optimal groundwater depth

The privately optimal groundwater depth for agriculture can be calculated by setting the first derivative of the agricultural production value with respect to the groundwater depth equal to zero (equation (3.7)). The agricultural production value is the sum of the production values of the various crops, which can be calculated by multiplying crop yields, prices and areas:

$$\frac{\mathrm{d}\left(\sum_{i=1}^{i=m} Y_i P_i A_i\right)}{\mathrm{d}D} = 0 \tag{3.7}$$

c) Relationship between the nature index and groundwater depth

Van Beusekom et al. (1990) show that the conservation values of plant species decline exponentially with declining groundwater depths. More recently, Witte (1998) attempted to relate water management to the value of nature. Both studies show that the relationship between the value of nature and the groundwater depth depends on the composition of the vegetation. Changes in the groundwater depth have a larger impact on the value of vegetation dominated by 'wet' species on loamy soils than on vegetation dominated by 'dry' species on sandy soils. The nature index N reflects the relationship between the conservation value of ecological benefits and the groundwater depth. The slope of the nature index $-s$ depends on the composition of the vegetation. It has a negative value, which means that the nature index decreases with deeper groundwater depths:

$$N(D) = \alpha e^{-sD} \qquad \text{for } D \geq 0.2 \text{ m bss} \tag{3.8}$$

The first damage to nature can be observed if the average groundwater depth in spring becomes lower than 0.2 m bss (N=1 for $D = 0.2$ m). The nature index will be determined on the basis of the observations derived from relationships presented by van Beusekom et al. (1990). In order to calculate the monetary value of ecological benefits, the nature index is multiplied by an annual monetary value of nature V for a nature index equal to one: $N=1$.

d) Socially optimal groundwater depth

The social optimum can be derived by equalising the absolute values of the first derivatives of the agricultural production value and the ecological benefits of the agricultural nature management function with respect to the groundwater depth (equation (3.9)).

$$\left| \frac{\mathrm{d}(\sum_{i=1}^{i=m}(Y_i P_i A_i)}{\mathrm{d}D} \right| = \left| \frac{\mathrm{d}(NVA_\mathrm{n})}{\mathrm{d}D} \right| \tag{3.9}$$

According to the Neo-Paretian criterion, there are allocation profits if the absolute value of benefits to nature (i.e. change in the groundwater depth from D^* to D^{**}) exceeds the absolute value of losses to agriculture (equation (3.10)). Winners are able to compensate losers if the following condition is fulfilled:

$$\left| g(D^{**})VA_\mathrm{n} - g(D^*)VA_\mathrm{n} \right| > \left| \sum_{i=1}^{i=m} f_i(D^{**})P_i A_i - \sum_{i=1}^{i=m} f_i(D^*)P_i A_i) \right| \tag{3.10}$$

e) Sensitivity analysis

The results of the model depend particularly on the characteristics of the nature index and the annual monetary valuation of nature. In order to analyse the impact on the results, the sensitivity to various values of the following three parameters is assessed. First, the effect of another composition of the vegetation on the results is studied. Secondly, the extent to which different annual monetary values of nature V influence the results is tested. Finally, the impact of changes in the relative share of agricultural area with special ecological value in total agricultural area on the results will be investigated: A_n/A.

3.3 Application to the eastern cattle-raising area

The model is applied in an empirical analysis to a desiccated study area. The eastern cattle-raising area in the Netherlands (see Figure 3.1) is chosen because in this area agriculture and nature compete with respect to the shallow groundwater depth.

Figure 3.1 Location of the eastern cattle-raising area in the Netherlands.

The main agricultural activities in the eastern cattle-raising area are dairy farming and cattle breeding. The total area comprises 300,000 hectares and consists mainly of sandy podzol soil. The crops considered in the analysis are grass, maize, potato, sugar beet and grain. The cropping pattern, crop yields and crop prices are shown in Table 3.1. These crop yields are assumed to be maximum yields, because we suppose that they are harvested at agriculturally optimal groundwater depths. This assumption seems reasonable, since in the second half of the twentieth century groundwater depths were lowered in order to create favourable crop-cultivation conditions. The optimal groundwater depth for grassland is 0.6 m bss and for arable crops 1 m bss (in which case there are no losses).

Table 3.1 Cropping pattern, physical crop yields and prices in the eastern cattle-raising area in 1992.

	Grass	Maize	Potato	Sugar beet	Grain
Cropping pattern (%)	60	25	5	5	5
Physical crop yields (tonnes/ha)	61.3	43.0	46.0	60.9	4.6
Price (€/tonne)	21.2	38.7	52.1	45.1	208.7

Source: LEI-Dutch Farm Accountancy Data Network, which is based on a representative sample of farms.

The relationships between crop yields and groundwater depth for five crops are estimated for quadratic and cubic functional forms. The ordinary least squares method is used. The regression is based on 51 observations of crop yields for groundwater depths varying between 0.1 and 2.6 m bss, with an interval of 0.05 m. These yields are derived from the yield losses tables developed by Working group HELP (1987), presented in Table 3.2. In order to obtain a complete data set (51 observations), percentages that are not available from the yield losses tables are calculated by linear interpolation. The complete data set used for the estimation is given in Oltmer (1999). The data for the regression analysis describing the impact of the groundwater depth on physical yields of grass and arable crops are derived from yield losses tables (code H2b, number 61 of Table G7 and B7) for sandy podzol soil developed by Working group HELP (1987).

Table 3.2 Yield losses percentages (in parenthesis minimum and maximum crop yields *Y* in tonnes/ha in 1992).

Groundwater depth (m)	Yield losses (%) [b]					
	Grassland	Arable land	Maize [a] Z=X	Potato [a] Z=1.15X+0.5	Sugar beet [a] Z=0.85X-0.5	Grain [a] Z=1.05X-2.5
0.10	21	31	31(*Y*=29.7)	31(*Y*=31.7)	31(*Y*=42.0)	31(*Y*=3.2)
0.15	17	27	27	27	27	27
0.20		23	23	23	23	23
0.25	10	15	15	15	15	15
0.30	5					
0.35	2	9	9	9	9	9
0.50	0	6	6	6	6	6
0.60	0 (*Y*=61.3)	2	2	2	2	2
0.70	1					
0.75	1					
1.00		0	0 (*Y*=43.0)	0 (*Y*=46.0)	0 (*Y*=60.9)	0 (*Y*=4.6)
1.05	3	X=4	Z=4	Z=5.1	Z=2.9	Z=1.7
1.10	4	X=4	Z=4	Z=5.1	Z=2.9	Z=1.7
1.40	8	X=9	Z=9	Z=10.85	Z=7.15	Z=6.95
1.50	10	X=12	Z=12	Z=14.3	Z=9.7	Z=10.1
1.70	15	X=16	Z=16	Z=18.9	Z=13.1	Z=14.3
2.00	21	X=23	Z=23	Z=26.95	Z=19.05	Z=21.65
2.60	25(*Y*=46.0)	X=26	Z=26	Z=30.4	Z=21.6	Z=24.8

a) Drought damage losses (Z) for the different arable crops are derived from the average yield losses of arable crops (X) on t he basis of given crop and soil-specific formulas (Working group HELP, 1987, Table 8).

b) Since precipitation and evaporation in the study area are comparable to precipitation and evaporation in the reference district, the yield losses do not need to be corrected (Working group HELP, 1987, p.29).

The yield losses tables of Working group HELP (1987) are commonly used to determine changes in agricultural crop yields as a result of land consolidation projects. They show wet and drought damage percentages for various groundwater depths. These percentages are soil-type-specific and are given for grassland as well as arable land. Drought damage of the different arable crops can be derived from the average yield losses of arable land on the basis of given crop- and soil-specific formulas (Working group HELP, 1987, Table 8).

As precipitation and evaporation vary per district, correction factors have to be used to adjust yield losses figures per district (Working group HELP, 1987, p. 29). Only those factors that lead to a decrease in production are considered in the yield losses tables (Feddes and van Wijk, 1990). Production losses due to e.g. a decrease in the workability of the soil are considered. Changes in farm management costs, e.g. lower harvesting cost or higher machinery cost due to a decline in the carrying capacity of the soil, are not considered. Losses are based on hydrological model simulations, field experiments and expert judgement, using precipitation and evaporation patterns over a period of 30 years. The losses given are an average for a number of years. The impact of weather effects (from year to year and within years) is therefore implicitly taken into account.

Statistics of the data derived from the yield losses percentages presented in Table 3.2 are shown in Table 3.3.

Table 3.3 The mean, standard error of mean and standard deviation of the data used.

	Grass	Maize	Potato	Sugar beet	Grain
Mean of crop yields (tonnes/ha)	54.2	37.1	38.8	53.7	4.0
Standard error of mean (tonnes/ha)	0.76	0.57	0.70	0.70	0.06
Standard deviation of crop yields (tonnes/ha)	5.5	4.1	5.0	5.0	0.4

Source: Based on 51 observations derived from the yield losses tables of Working group HELP (1987).

The observations describing the impact of groundwater depths on the conservation value are derived from the vegetation- and soil-type-specific relationships presented by van Beusekom et al. (1990). Two sets of observations are used to show the sensitivity of the results to another composition of the vegetation. The first set relates to vegetation dominated by ‘dry’ species on sandy soils, and the second set to vegetation dominated by ‘wet’ species on loamy soils. For both sets of observations there is indeed no damage (N=1), at a groundwater depth of 0.2 m bss. For the first set of observations, half the total value of nature is lost (N=0.5) at a groundwater depth of 0.9 m bss. Only one-quarter remains (N=0.25) at 1.6 m bss. For the second set of observations, half the total value of nature is lost (N=0.5) at a groundwater depth of 0.55 m bss and only one-quarter remains (N=0.25) at 0.9 m bss. These observations show that vegetation dominated by ‘wet’ species on loamy soils is indeed more sensitive to deeper groundwater depths than vegetation dominated by ‘dry’ species on sandy soils.

The monetary value of nature is not easy to establish because many of the provided goods and services are not marketed. Several techniques for economic valuation of nature have been developed (e.g. contingent valuation method, travel cost method, hedonic pricing method). Studies present different monetary values of nature (see Costanza et al., 1997). The public benefits of agricultural nature management on Dutch peat meadow land were estimated by means of the contingent valuation method by Brouwer and Slangen (1998), who found a value of about €1,600 ha^{-1} yr^{-1}. Since this value is derived for peat soil rather than sandy podzol soil, benefit transfer is questionable. However, because of lack of a more accurate estimation, we use €1,600 ha^{-1} yr^{-1} for the monetary value of nature V as a benchmark in our analysis and assess the sensitivity of the results to another monetary value of nature, namely €1,000 ha^{-1} yr^{-1}.

Finally, we assume that the size of the agricultural area with an ecological value is one-quarter of the total agricultural area in our study area, because about 20-25% of total agricultural land in the Netherlands has special ecological value (Slangen, 1992). We use this share as a reference. The impact of changes in this share on results is assessed in a sensitivity analysis.

3.4 Optimal groundwater levels

The results of the application of the model framework to the study area are presented in this section. The observations and estimated quadratic and cubic functions are shown in Figure 3.2 for the yield of grass.

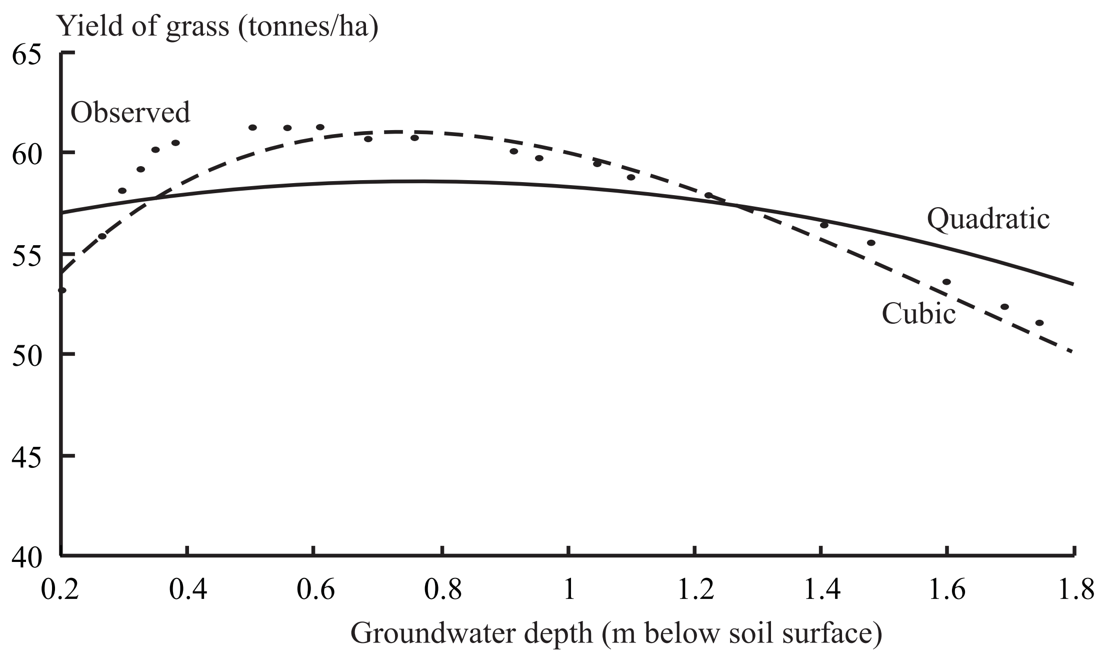

Figure 3.2 Observed and estimated annual yield of grass as a function of the groundwater depth.

The cubic functional form (equation (3.6)) describes the observed data better than the quadratic functional form (equation (3.5)) does. A theoretical disadvantage of the cubic form is that multiple optima are possible, but this is no problem for the domain analysed. It is interesting to note that the negative impact of wet damage on crop yield is larger than the negative impact of drought damage (the left part of the curve is steeper than the right part).

On the basis of the graphic representation, the significance of the values of all estimated parameters and the values of the adjusted R^2, which are higher for the cubic than for the quadratic functions, the cubic functional form is preferred to the quadratic functional form. Estimation results of cubic functional forms for the five crops are shown in Table 3.4.

Table 3.4 Estimation results of the relationship between annual crop yields and groundwater depth for the cubic functional form (equation (3.6)) (with the t-values in parentheses).

Parameter	β_0	β_1	β_2	β_3	Adjusted R^2
Grass	47,503 (70.62)	396.0 (18.93)	-3.38 (-19.06)	0.007 (16.50)	0.97
Maize	27,229 (64.37)	410.2 (31.19)	-3.27 (-29.33)	0.007 (25.18)	0.98
Potato	28,823 (62.65)	456.8 (31.93)	-3.73 (-30.78)	0.008 (26.58)	0.98
Sugar beet	38,936 (65.07)	556.7 (29.92)	-4.32 (-27.38)	0.009 (23.33)	0.97
Grain	2,885 (75.36)	44.1 (37.07)	-0.34 (-34.06)	0.0007 (28.77)	0.98

Optimal groundwater depths and related yields of the cubic functions are shown in Table 3.5.

Table 3.5 Optimal groundwater depths and the related maximum yields of the cubic functions.

	Grass	Maize	Potato	Sugar beet	Grain
Optimal groundwater depth (m bss)	0.75	0.89	0.84	0.90	0.88
Maximum yields (tonnes/ha)	61.0	42.9	45.6	60.6	4.6

Figure 3.3 shows that under prices of 1992 no changes in cropping pattern could be expected in the analysis as a result of adjustments in the groundwater depth. The gross production value per hectare of a certain crop will not become higher or lower than that of another crop if the groundwater depth changes, as illustrated by the fact that curves do not intersect. However, the gross production values of crops might change when product prices change, in which case changes in groundwater depths might affect cropping patterns.

The relationship (equation (3.11)) between the agricultural production value *APV* (€/ha) and groundwater depth is derived by regression of calculated agricultural production values on groundwater depths. The calculated agricultural production value is the sum of the estimated production value of each crop times the fraction of each crop in the cropping pattern.

$$APV = 1063.569 + 11.942D - 0.098D^2 + 0.0002D^3 \tag{3.11}$$

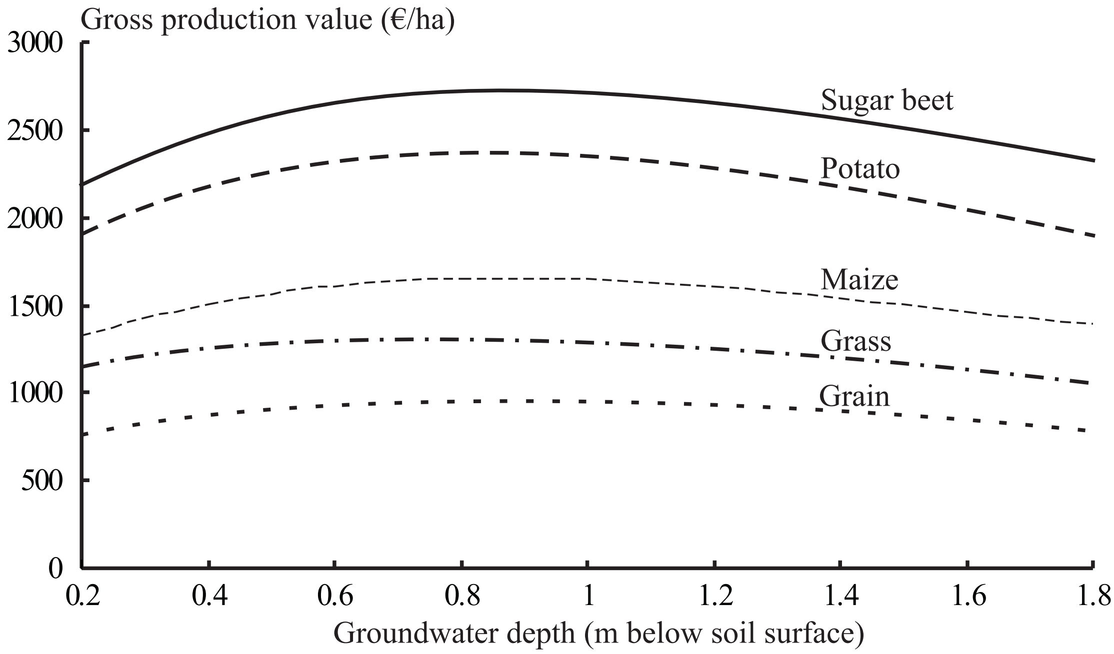

Figure 3.3 Relationship between gross production values and groundwater depth.

The relationship between the nature index and groundwater depth (equation (3.8)) has been determined on the basis of both sets of observations derived from van Beusekom et al. (1990). For vegetation dominated by 'dry' species on sandy soils, α=1.2 and s=1. For vegetation dominated by 'wet' species on loamy soils, α=1.5 and s=2. Figure 3.4 shows that vegetation dominated by 'wet' species on loamy soils is more sensitive to changes in groundwater depths (the slope of the index is steeper) than 'dry' species on sandy soils.

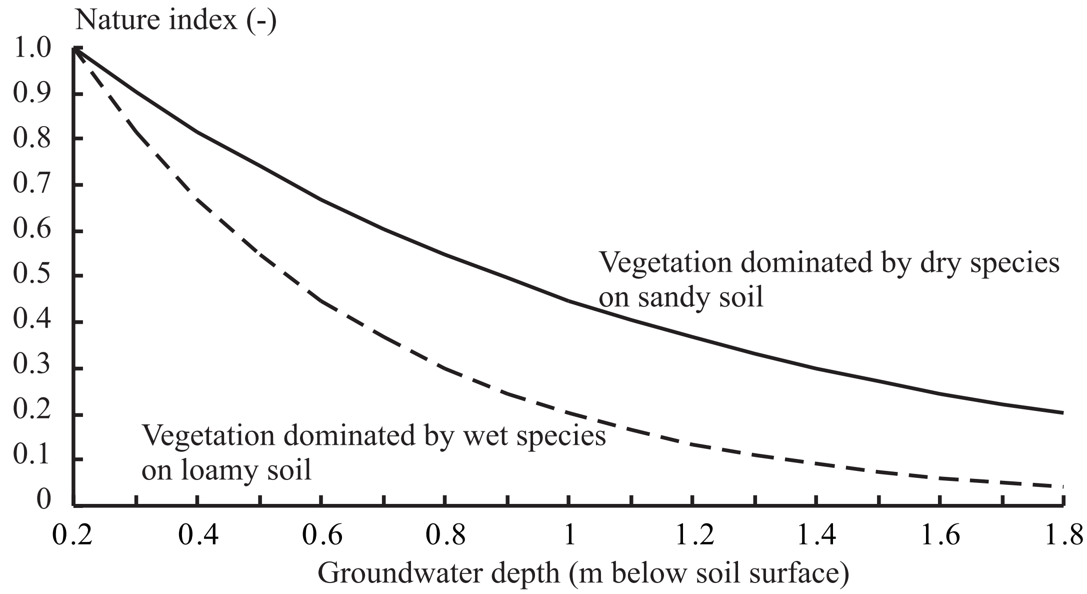

Figure 3.4 Nature index as a function of the groundwater depth for vegetation dominated by 'dry' species on sandy soils and for vegetation dominated by 'wet' species on loamy soils.

The agricultural production value and monetary value of ecological benefits of 4 ha of farmland, both as a function of the groundwater depth, are presented in Figure 3.5. The social value is the sum of the agricultural production value of these 4 ha and the value of ecological benefits of 1 ha of the 4 ha, since only one-quarter has an ecological value.

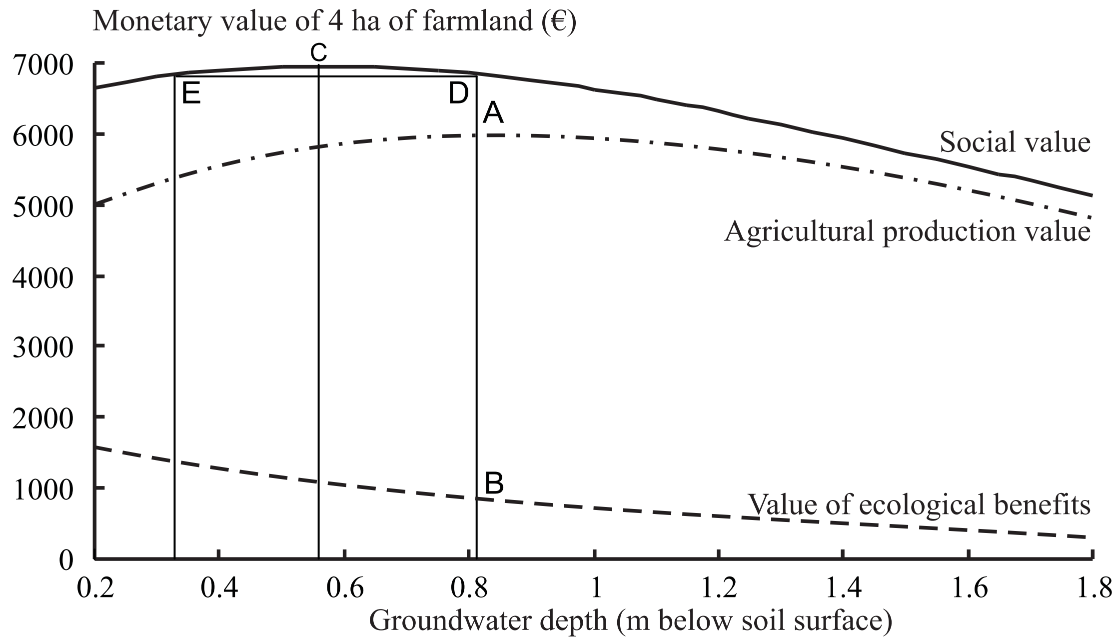

Figure 3.5 The social value, agricultural production value and value of ecological benefits (for *V*=1600 and vegetation dominated by 'dry' species on sandy soils) of 4 ha of farmland.

The agriculturally optimal groundwater depth is 0.81 m bss. The associated agriculturally optimal production value is €1,494.2 per hectare, which is equivalent to €5,976.7 for 4 ha (see point A in Figure 3.5). At the agriculturally optimal groundwater depth, the monetary value of ecological benefits is €868.8 (see point B).

Figure 3.5 shows that there will be losses to agriculture and nature if groundwater depths are deeper than the agricultural optimum, which is not attractive for either interest. If groundwater depths are less deep than the agricultural optimum, there will be losses to agriculture but benefits to nature.

The socially optimal groundwater depth is 0.57 m bss. The associated socially optimal value is €6,956.6 (see point C in Figure 3.5), of which €5,846.7 is for agriculture and €1,109.9 for nature. Annual agricultural losses resulting from the shift from a private to a social optimum are €130 for 4 ha and annual benefits to nature are €241.1.

At the agriculturally optimal groundwater depth of 0.81 m the social value of 4 ha of farmland is €6845.5 (see point D). At a groundwater depth of 0.34 m, the same social value could be achieved (see point E). It turns out that the distribution of welfare between agriculture and nature can be different for the same social value. Any groundwater depth between 0.34 and 0.81 m increases the social value compared with the agricultural optimum. Within this range the absolute value of benefits to nature exceeds the absolute value of losses to agriculture (equation (3.10)) and there are allocation profits according to the Neo-Paretian criterion. The winners are able to compensate the losers and the two parties may negotiate about compensation payments. Whether real compensation has to be paid depends on the assignment of rights to lower groundwater depths, reaching agreement and policy decisions.

At the social optimum the annual marginal losses and benefits to agriculture and nature from changing the groundwater depth by 1 cm at that point (equation (3.9)) are €11.1 (see point A in Figure 3.6). If the social planner were to introduce a charge system, the size of these annual marginal losses and benefits gives an indication of the level of charges farmers would have to face in order to restrain them from lowering the groundwater depth by 1 cm. The annual charges farmers would have to face in order to restrain them from changing the groundwater depth from a social to a private optimum (by 0.24 m) would have to at least equal to potential annual benefits to agriculture. In our case study, these annual benefits amount to €130 for 4 ha.

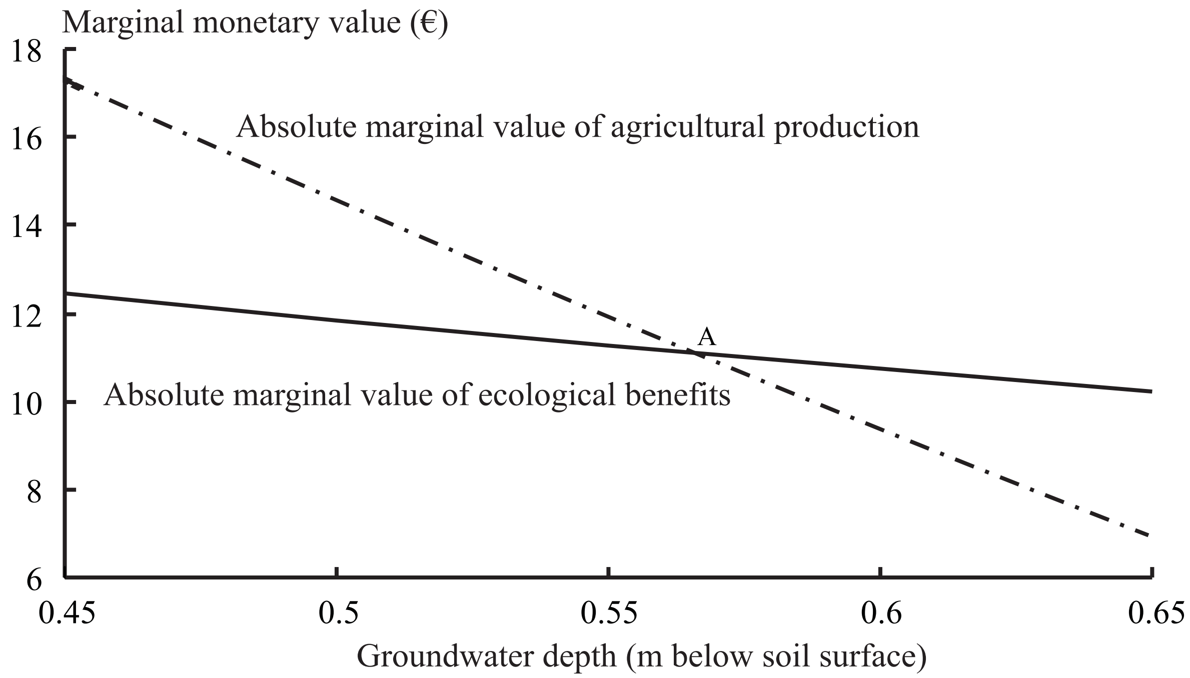

Figure 3.6 First derivative of agricultural production value function and value of ecological benefits function.

Theoretically, charges could be used as a tool to achieve the social optimum. In practice it would be complicated for the following reasons. Firstly, it is hard to define a good base for a charge. A charge on a change in the groundwater level is complicated, because external factors also affect the groundwater level. Secondly, location-specific circumstances should be taken into account, there is heterogeneity in the vulnerability of areas to changes in groundwater levels. Thirdly, groups of farmers will be affected by a change in the groundwater depth; it is hard to exclude farmers. Fourthly, tax revenues can be small compared with transaction costs. Finally, farmers seem to be risk-averse with respect to wet damage and the impact of a charge on water demand may consequently be small.

The impact of another composition of the vegetation and of another annual monetary value of nature on socially optimal outcomes is shown in Table 3.6. As the value ascribed to nature increases, the groundwater level will be more attuned to nature. Socially optimal groundwater depths vary in the sensitivity analysis between 0.66 and 0.32 m bss. Annual agricultural losses resulting from the shift from a private to a social optimum vary between €44.7 and €578.4 for 4 ha. This is 0.7-9.7 % of the agricultural optimum production value. Annual benefits to nature resulting from this shift range between €85.5 to €796. The increase in social value varies between €40.8 and €217.6 for 4 ha.

Table 3.6 Results of the sensitivity analysis of the impact of another composition of the vegetation and of another annual monetary value of nature on socially optimal outcomes (annual values in € for 4 ha of farmland; 1 ha of these 4 ha of farmland also has special ecological value).

Vegetation dominated by	'dry' species on sandy soils	'dry' species on sandy soils	'wet' species on loamy soils	'wet' species on loamy soils
Annual monetary value of nature (V)	(ref.) 1,600	1,000	1,600	1,000
Optimal groundwater depth (m)	0.57	0.66	0.32	0.61
Agricultural production value	5,846.7	5,932.0	5,398.3	5,892.9
Value of ecological benefits	1,109.9	628.5	1,267.8	438.4
Social value	6,956.6	6,560.5	6,666.1	6,331.3
Losses to agriculture [a)]	130.0	44.7	578.4	83.8
Benefits to nature [a)]	241.1	85.5	796.0	143.6
Marginal losses/benefits, per cm [b)]	11.1	6.3	25.4	8.8

a) Losses and benefits in the social optimum compared with the agricultural optimum.
b) Marginal losses and benefits to the social optimum from changing the groundwater depth by 1 cm.

Figure 3.7 shows the results of the sensitivity analysis performed to analyse the impact of changes in the relative share of agricultural area with special ecological value in total agricultural area A_n/A on the socially optimal groundwater depth. A constant monetary value of nature V is used, although the marginal value will change with the total size of nature in a country. In our analysis a lower monetary value of nature has a similar impact on socially optimal groundwater depths as a decrease in the relative share of area with special ecological value. In order to isolate the impact of changes in the relative share of area with ecological benefits, we use a constant marginal monetary value of nature.

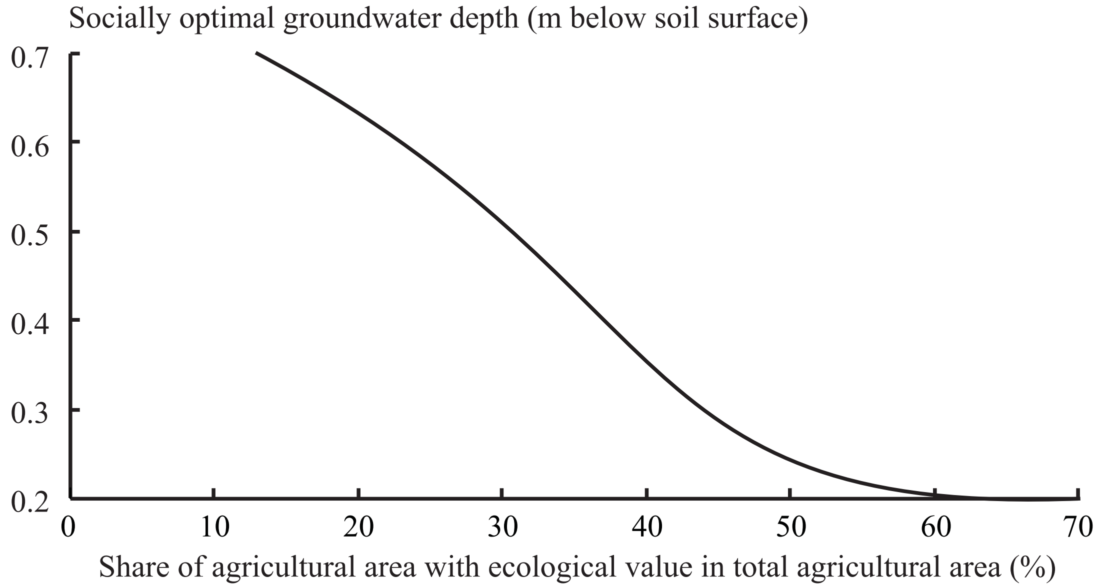

Figure 3.7 Socially optimal groundwater depth as a function of the relative share of agricultural area with special ecological value in total agricultural area A_n/A.

It turns out that the relative share of area with ecological value is important for the socially optimal groundwater depth (Figure 3.7). If 70% of the agricultural area has special ecological value, the groundwater depth will be fully attuned to nature (0.2 m bss). Agricultural losses will in that case be 15% of the agricultural optimum production value.

Costs to agriculture are smaller than estimated due to second-order effects, like changes in farm management practices. Especially changes in cropping pattern seem to be promising. Wet-tolerant crops or new crops, e.g. the energy crop willow, can be cultivated. It is therefore recommended for future research to consider second-order effects in the analysis.

The model developed in this chapter is not only useful for determining the optimal groundwater depth, but can also be used for sensitivity analysis. The optimal groundwater depth can be identified under various assumptions regarding values of nature as well as prices of agricultural commodities. The cost-benefit analysis can be used as a starting point for a more general economic analysis that will identify how optimal behaviour can change under various assumptions regarding values of nature. It is therefore recommended for future research to extend the analysis. Since there is uncertainty about the value of nature, it might also be interesting to introduce mechanisms that look at outcomes under a probability of values of nature and then compute an optimal solution based on an average value.

3.5 Conclusions

In this chapter a model is developed to analyse the inter-sectoral trade-off between agricultural production values and the monetary value of ecological benefits as a result of changes in the groundwater depth. Cost-benefit analysis is applied to provide insight into the benefits and losses to nature and agriculture resulting from a shift from privately to socially optimal groundwater depths. This model has been illustrated by an empirical analysis in a study area, which identifies existing gaps in knowledge. Although the results are illustrative, some general conclusions can be drawn.

The study shows that agricultural production losses for achieving the socially optimal groundwater depth are higher for vegetation dominated by 'wet' species on loamy soils than for vegetation dominated by 'dry' species on sandy soils. Also, losses will increase if society attaches a higher annual monetary value to nature and if the relative share of agricultural areas with special ecological value in the total agricultural area becomes larger. The study also shows that agricultural wet-damage losses are more serious than drought-damage losses.

Theoretically, charges on lowering the groundwater depth could be used as a tool to achieve the social optimum, although in practice it would be complicated. Compensating payments could be used to redistribute welfare – the costs and benefits – among the affected parties.

Finally, insight into losses and benefits to agriculture and nature can support policy decisions concerning the introduction of economic incentives or other policy instruments into water management in order to restore desiccated ecosystems in agricultural areas.

4 Non-uniform groundwater level management in agricultural areas and nature reserves[5]

4.1 Introduction

In the Netherlands most nature reserves are small, often covering less than 1 km^2. As a result, they are strongly dependent on water management in adjacent farmland. Higher groundwater levels restore desiccated nature reserves, but may cause unintended wet damage to adjacent farmland. Despite several attempts to analyse the impact of policies aimed at reducing the desiccation of nature, there is still a lack of insight into the cost-effectiveness of restoration projects.

To analyse restoration projects from an ecohydrological point of view one usually uses a number of interrelated complex models (Witte et al., 2001). First, the impact of measures on both the groundwater level and the seepage flux are studied by means of a steady-state geo-hydrological model. Then, the dynamic character of the groundwater level and the availability of soil moisture are calculated by means of a model for the unsaturated zone. This hydrological model generates the input to an ecological model, which finally computes the ecological effect of the measures.

This approach has a number of shortcomings. Firstly, there is usually a lack of data (especially geographical data) to feed the models. Secondly, the approach is very time-consuming. Thirdly, because of its complexity and the large amount of required input data, the approach is not transparent and may easily contain errors, for instance concerning the connection between the models. Finally, the models are often so complex that in practice only a limited number of measures or scenarios can be assessed. A relatively transparent, multidisciplinary economic-hydrological model, which is easy to manage, may overcome these shortcomings.

The aim of this chapter is to develop a prototype of such a multidisciplinary and user-friendly model. Our model will be designed to compare agricultural costs with benefits to nature of restoration projects that reduce the desiccation of nature. The model provides

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[5] Chapter 4 is a modified version of Hellegers P.J.G.J. and J.P.M. Witte, forthcoming, An economic ecohydrological approach for allocation of compensation payments for the restoration of desiccated nature reserves. *Ecohydrology and Hydrobiology*, 00(0):000-000.

insight into the cost-effectiveness of restoration projects, by showing the costs of full and partial restoration. This decision-support model can be applied to evaluate subsidy requests for nature conservation. We will demonstrate the use of the model to allocate payments as compensation for agricultural losses due to the restoration of nature, by applying it to three characteristic nature reserves. The model will also be used to compare investment costs of hydrological restoration measures to re-wet nature, which achieve the same goal.

In this chapter the first and second module of Figure 1.5 will be studied: the hydrological and economic modules. The basic features of the model, including the kind of output it generates in the form of benefits to nature and costs to agriculture, are described in Section 4.2. Section 4.3 discusses allocation possibilities of compensation payments for restoration of desiccated nature among three characteristic reserves. Section 4.4 provides insight into (a) the cost-effectiveness of restoration in nature reserves themselves; (b) the possible division of payments among the characteristic reserves; and (c) the cost-effectiveness of hydrological measures to re-wet nature. Section 4.5 contains the discussion and conclusions.

4.2 A new economic-hydrological approach

Our prototype model consists of four modules, which are linked to each other as schematically represented in Figure 4.1.

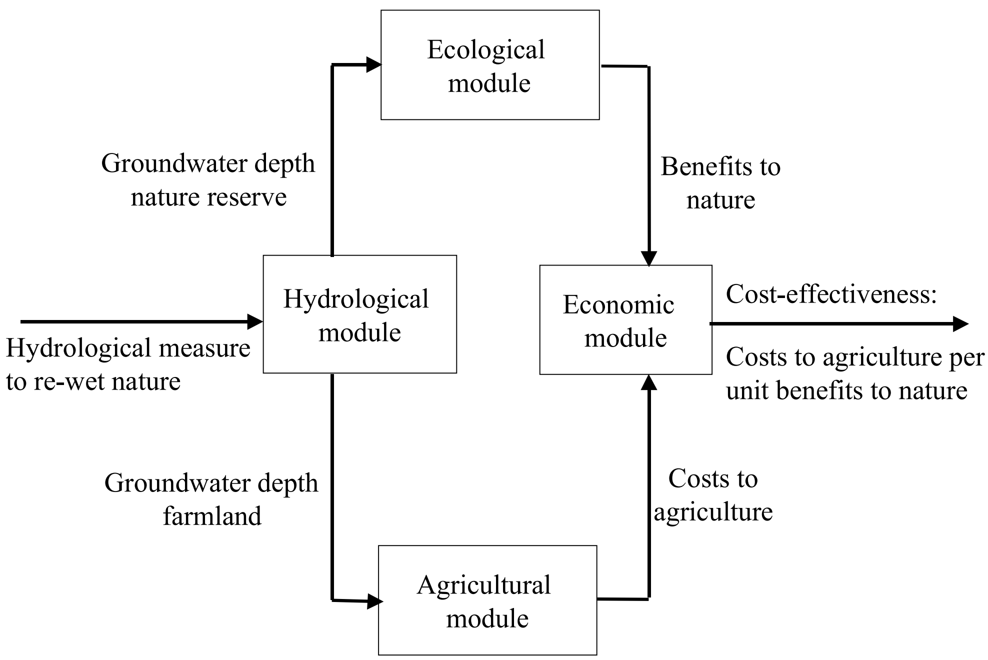

Figure 4.1 Schematic representation of the model. An arrow represents input to/output from a module.

The *hydrological module* simulates the impact of hydrological measures to re-wet nature on the groundwater depth in the nature reserve and in adjacent farmland. The groundwater depth determines the characteristics of the environment of the vegetation and crop, in particular the availability of soil moisture and oxygen in the root zone. It is therefore input to both the *ecological and agricultural module*, which determine ecological benefits and agricultural costs as a result of changes in the groundwater depth. The *economic module* provides insight into the cost-effectiveness of restoration projects that reduce the desiccation of nature on the basis of these benefits to nature and costs to agriculture. In this section, each of these modules is explained separately and in more detail.

Economic module

The economic module provides insight into the cost-effectiveness of nature restoration on the basis of benefits to nature and costs to agriculture due to higher groundwater levels.

Benefits to nature can be expressed in various ways. In this study we will use two indicators: (1) an annual increase in the unaffected surface area of the nature reserve ΔA^{u} (ha/yr); and (2) an annual increase in the ecological conservation value of the nature reserve ΔV^{r} (in conservation value units per year; abbreviated cvu/yr). The first is an appealing measure, because it fits the goal of the Dutch government to reduce the desiccated area by 40% in the year 2010 as compared to the situation in 1985, and because it is an objective measure. The second indicator, which is in essence subjective, focuses on ecological quality as measured by the ecological conservation value. It better reflects the value mankind ascribes to nature.

The increase in costs to agriculture ΔC (€/yr) due to higher groundwater levels are in this study equal to the sum of the annual change in agricultural wet damage costs ΔC^{w} (€/yr) and drought damage costs ΔC^{d} (€/yr) to crops. The first effect is usually more serious than the latter effect, which means that costs to agriculture increase if the groundwater level is raised.

Two cost-effectiveness indicators are used, since benefits to nature are defined in two ways. The first indicator E^{A} is defined as an increase in costs to agriculture per unit increase in the unaffected surface area (equation (4.1)) (in €/ha). The second indicator E^{V} is defined as an increase in costs to agriculture per unit increase in the ecological conservation value (equation (4.2)) (in €/cvu).

$$E^{\mathrm{A}} = \frac{\Delta C}{\Delta A^{\mathrm{u}}} \tag{4.1}$$

$$E^{\mathrm{V}} = \frac{\Delta C}{\Delta V^{\mathrm{r}}} \tag{4.2}$$

Capital losses to landowners due to a lower value of land as a result of wet damage are not considered in the analysis, because it is reasonable to assume that the value of land does no decline if farmers are compensated for wet damage. Investment costs of hydrological measures to re-wet nature are at first not considered in both indicators, since we assume that such adjustments in groundwater levels can be realised by means of existent infrastructure. The investment costs of hydrological measures to re-wet nature, which require substantial investment, will be considered in the cost-effectiveness analysis at the end of Section 4.4.

Hydrological module

In the typical setting of water management in the Netherlands, the groundwater level is traditionally determined at the optimal level for agriculture. Raising the groundwater level can alleviate the desiccation problem. We will use the model to study two kinds of hydrological measures to re-wet nature: (1) an increase in the groundwater recharge rate, e.g. by inlet of foreign water; and (2) the creation of a hydrological buffer zone between the farmland and the nature reserve. The hydrological module simulates the impact of these measures on the groundwater table in the nature reserve and its adjacent farmland. It is designed for a specific setting of nature and agriculture. As many nature reserves in the Netherlands are abandoned rectangular agricultural parcels, we assume that our nature reserve is elongated with farmland on both sides. Figure 4.2 gives a picture of our hydrological module.

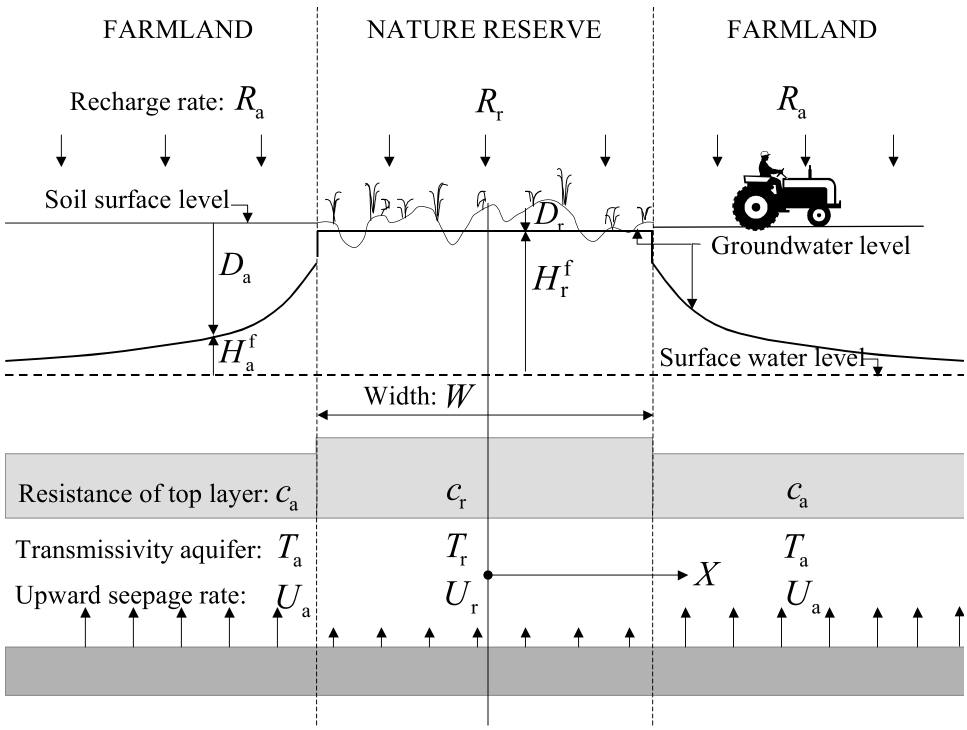

Figure 4.2 Hydrological schematisation of the elongated nature reserve and its adjacent farmland.

The subsoil of the reserve and adjacent farmland consists of a resistance layer on top of an aquifer. A subscript 'a' refers to agriculture and a subscript 'r' to nature. For a complete mathematical description and the underlying assumptions, see Appendix A.

We make a distinction between groundwater *level* H^{f} and groundwater *depth* D. H^{f} refers to the height of the groundwater table relative to the surface water level of the farmland. Spatial differences in H^{f} determine the vertical and horizontal flows of groundwater. D refers to the height of the groundwater table relative to the soil surface. Groundwater depth is equal to the soil surface level minus the groundwater level (equations (A.13) and (A.14)). Our model computes an average groundwater level in the reserve $H_{\mathrm{r}}^{\mathrm{f}}$ (equation (A.8)), because the exact place of inundation differs considerably per reserve and ponding (formation of small areas of water) can not be modelled in an analytical manner. The groundwater level in farmland $H_{\mathrm{a}}^{\mathrm{f}}$ (equation (A.11)), on the other hand, decreases exponentially with an increasing distance from the reserve. Both $H_{\mathrm{r}}^{\mathrm{f}}$ and $H_{\mathrm{a}}^{\mathrm{f}}$ depend on the groundwater recharge rate R, the width of the nature reserve W and geohydrological characteristics: the hydraulic resistance of the top layer c, the transmissivity of the underlying aquifer T, and the intensity of the regional upward seepage rate U.

The soil surface of the farmland is considered to be flat, which is the typical situation in the Netherlands. In contrast to farmland, nature reserves are usually gently sloping. To account for this, we assume the soil surface level of the reserve to be normally distributed (equation (A.15)). As we work with an average groundwater level in the nature reserve, the groundwater depth in the nature reserve is normally distributed as well (equation (A.16)).

Ecological module

The ecological module assesses the ecological benefits of higher groundwater levels in reserves. To quantify benefits to nature, a hydrological base situation and an optimal situation have to be defined. The base situation refers to the current groundwater level in the nature reserve. The optimal situation is based on a historical hydrological reference, without desiccation of nature. The groundwater level of the reserve in 1950 is used as a reference, since it refers to the situation before large-scale land consolidation projects were carried out. This situation is often used as a reference for a former, less degraded state (e.g. Jansen and Runhaar, 2000). Hydrological restoration, studied here, is in practice not identical to full ecological restoration, which is a more gradual long-term process than hydrological restoration and can only be achieved if there is no irreversible damage. It is also important to note in this respect that ecological restoration is only sustainable if measures are maintained in the future. Turner et al. (2000) also developed an ecological-economic analysis.

The total surface area of the reserve A^t is partly affected (desiccated) and partly unaffected: ($A^t = A^a + A^u$). *The maximum increase in the unaffected surface area of the nature reserve* ΔA^u will be derived from the normally distributed groundwater depth in the base situation and that in the optimal situation, as shown in Figure 4.3. The overlap of both distributions shows the percentage of unaffected surface area, which increases as the groundwater level is raised. If the groundwater level increases by 0.3 m, the desiccated surface area in Figure 4.3 will be fully restored (both distributions fully overlap). The increase in the unaffected surface area ΔA^u is equal to the decrease in the desiccated (affected) surface area ΔA^a.

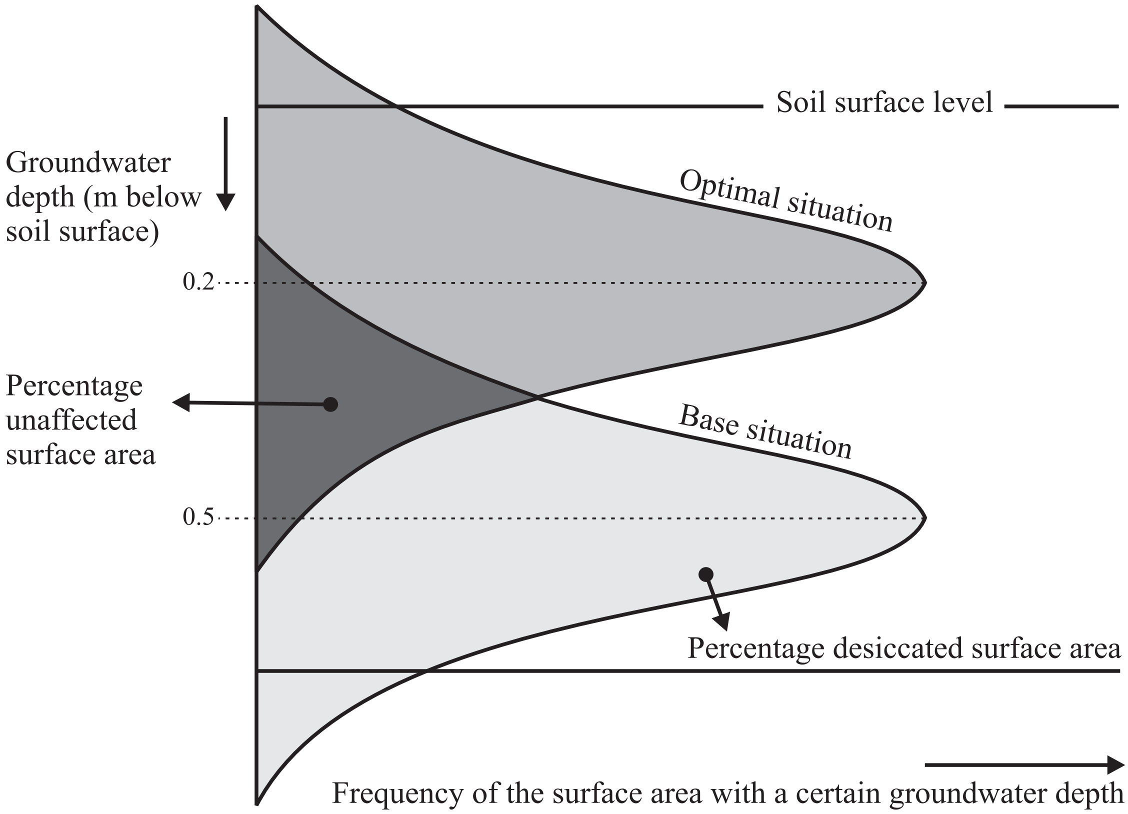

Figure 4.3 The normally distributed groundwater depth in the base situation and that in the optimal situation.

The increase in the conservation value of the nature reserve ΔV^r will be derived from the conservation value of the reserve in the optimal situation minus the conservation value of the reserve in the base situation. To calculate the conservation value we use the method developed for the ecohydrological model DEMNAT (Witte, 1998; van Ek et al., 2000).

We distinguish site types, which are spatially ecologically relevant entities. Table 4.1 shows the 20 site types used in this study. They are distinguished on the basis of three operational site factors that explain important differences in the species composition of the plant cover of the Netherlands: moisture regime (characterising water and oxygen regime), nutrient availability and acidity. These factors may directly or indirectly be affected by hydrological changes, which is why this classification of site types is very suitable for impact assessment studies on water management.

Table 4.1 Site types distinguished (after: Klijn et al., 1996).

		Nutrient-poor			Moderately nutrient-rich	Very nutrient-rich
		Acid	Neutral	Alkaline		
	Code	1	2	3	7	8
Water	1	11	12	13	17	18
Wet	2	21	22	23	27	28
Moist	4	41	42	43	47	48
Dry	6	61	62	63	67	68

For each of the three site factors a number of classes have been distinguished. The factor moisture regime for example, is divided into the classes water, wet, moist and dry. Combining these classes results in the site types of Table 4.1 (for instance, site type 21: wet, nutrient-poor and acid). The influence of acidity on the species composition is far less pronounced in nutrient-rich ecosystems than in nutrient-poor ecosystems. Acidity has therefore not been used as a classification characteristic in nutrient-rich ecosystems.

In our model, the conservation value of the nature reserve V^{r} is deduced from the conservation values of site types that occur in the nature reserve V^{s}. Table 4.2 shows the conservation values of site types that we will use for our model. The source of these values is as follows. Witte (1998) developed a method by which conservation values on a cardinal scale are ascribed to site types. His method is based on two assumptions: (1) the more rare a site type is (in national and international perspective), the higher its conservation value; and (2) an expansion of a site type in the Netherlands should be regarded as a positive development. Five experts in the field of botany, who were consulted single-blind, unanimously preferred Witte's valuation method to other existing valuation methods (Witte, 1998). The disadvantages of Witte's method are that it is rather complex, it requires accurate data about rarity and does not yield values for all site types that occur in the Netherlands. Therefore, Witte and Klijn (1997) developed another method: they formulated rules of thumb for the valuation of site types, based on the following three environmental policy goals of the Dutch government: to combat (1) desiccation; (2) eutrophication; and (3) acidification. With these three goals in mind, Witte and Klijn valued, for instance, wet site types higher than dry site types, nutrient-poor types higher than types rich in nutrients, and alkaline types higher than acid types. Table 4.2 gives the results of their formalised expert judgement. The policy-based expert values of Witte and Klijn (1997) appeared to correlate very well with the rarity-based values of Witte (1998). Witte and Klijn (1997) show a rank correlation coefficient of 0.97 for n=15.

Table 4.2 Conservation value units ascribed to the site types, based on Witte and Klijn (1997).

	Nutrient-poor			Moderately nutrient-rich	Very nutrient-rich
	Acid	Neutral	Alkaline		
Water	16	32	64	8	4
Wet	12	24	48	6	3
Moist	8	16	32	4	2
Dry	4	8	16	2	1

The operational site factors of Table 4.1 (moisture regime, nutrient availability and acidity) are conditioned by the more indirect factors: (1) groundwater depth; (2) soil type; and (3) occurrence of upward seepage. In contrast to the operational site factors, these conditioning factors are easy to obtain, e.g. from soil maps. For each combination of these three conditioning factors, Klijn et al. (1996) estimated the occurrence probabilities of site types (see also Klijn, 1997; Klijn et al., 1997). Table 4.3 shows, for instance, the occurrence probabilities $p_{k,j}$ of site types ($j=1,...,20$) on a humus sandy soil without lime and with upward seepage for five different classes of groundwater depth, the so-called Ground Water Table (*GWT*) classes ($k=1,...,5$). *GWT* classes are incorporated in every Dutch soil map.

Table 4.3 Occurrence probabilities of site types of each *GWT* class on a humus sandy soil without lime and with upward seepage (see Klijn et al., 1997, Appendix 4). Site types with an occurrence probability of zero have been omitted. Conservation values of each *GWT* class are also presented.

Ground Water Table class (*GWT*)		k=1	k=2	k=3	k=4	k=5
Groundwater depth in spring (m)		<0.3	0.30-0.55	0.55-0.65	0.65-1.0	>1.0
Occurrence probability of:	site type 12	0.111	0.088	0.031		
	site type 17	0.022	0.018			
	site type 21	0.316	0.259	0.031		
	site type 22	0.551	0.458	0.031		
	site type 41		0.088	0.453	0.127	0.030
	site type 42		0.088	0.453	0.025	
	site type 61				0.848	0.970
Conservation value of each *GWT* class V_k^g		20.75	19.19	13.00	4.81	4.12

This information can be used to predict changes in the occurrence probability of site types due to changes in the groundwater level. Table 4.3 shows the conservation values of each *GWT* class V_k^{g}, which is equal to the sum of occurrence probabilities $p_{k,j}$ times the conservation value scores of each site type V_j^{s}, shown in Table 4.2 (equation (4.3)).

$$V_k^{\mathrm{g}} = \sum_{j=1}^{j=20} p_{k,j} V_j^{\mathrm{s}} \tag{4.3}$$

The conservation value of the reserve V^{r} is equal to the sum of surface areas of each *GWT* class A_k^{t} times the conservation values of each *GWT* class V_k^{g}, shown in Table 4.3:

$$V^{\mathrm{r}} = \sum_{k=1}^{k=5} A_k^{\mathrm{t}} V_k^{\mathrm{g}} \tag{4.4}$$

Table 4.4 shows as an example surface areas in each *GWT* class and the conservation values of a reserve of 40 ha, consisting of a humus sandy soil without lime and with upward seepage (Table 4.3), for both the base and the optimal situation. In this example, the conservation value of the reserve increases by 91 cvu/yr when the optimal situation is assessed.

Table 4.4 Surface areas of each *GWT* class and the conservation values of a nature reserve of 40 ha.

	Surface area in each *GWT* class A_k^{t} (ha)					Conservation value of the reserve V^{r} (cvu/yr)
Situation	*k*=1	*k*=2	*k*=3	*k*=4	*k*=5	
Base	20.9	11.6	3.0	4.1	0.3	718
Optimal	34.2	4.6	0.6	0.5	0.0	809

Agricultural module

The agricultural module assesses the impact of changes in groundwater depths on production values. Agricultural wet (equation (4.5)) and drought damage costs (equation (4.6)) both depend on the following characteristics: (1) the length of the reserve L (because there is farmland on both sides, it is multiplied by 2); (2) fractions F_i of farmland with crop i (for $i=1,...,n$); (3) crop prices P_i; (4) potential optimal crop yields Y_i; and (5) changes in wet- and drought-damage fractions (w and d) of potential crop yields. Wet-damage fractions per crop w_i decrease with an increasing distance X from the middle of the reserve, since the groundwater depth in the farmland increases exponentially. Drought-damage fractions per crop d_i increase with increasing X.

$$C^{\mathrm{w}} = 2L \sum_{i=1}^{i=n} F_i P_i Y_i \int_{X=\frac{W}{2}}^{\infty} \Delta w_i(X)\, \mathrm{d}X \tag{4.5}$$

$$C^{\mathrm{d}} = 2L \sum_{i=1}^{i=n} F_i P_i Y_i \int_{X=\frac{W}{2}}^{\infty} \Delta d_i(X)\, \mathrm{d}X \tag{4.6}$$

Wet- and drought-damage fractions of potential physical crop yields at various groundwater depths are available from Working group HELP (1987). We fitted a continuous relationship through these discrete data; see Appendix B (equations (B.4) and (B.5)). Figure 4.4 shows the damage fractions of grass. It shows that wet damage is a decreasing convex function of groundwater depth between 0.2 and 0.75 m bss, whereas drought damage is an increasing convex function of groundwater depth within this range. For a complete description of the agricultural module, see Appendix B.

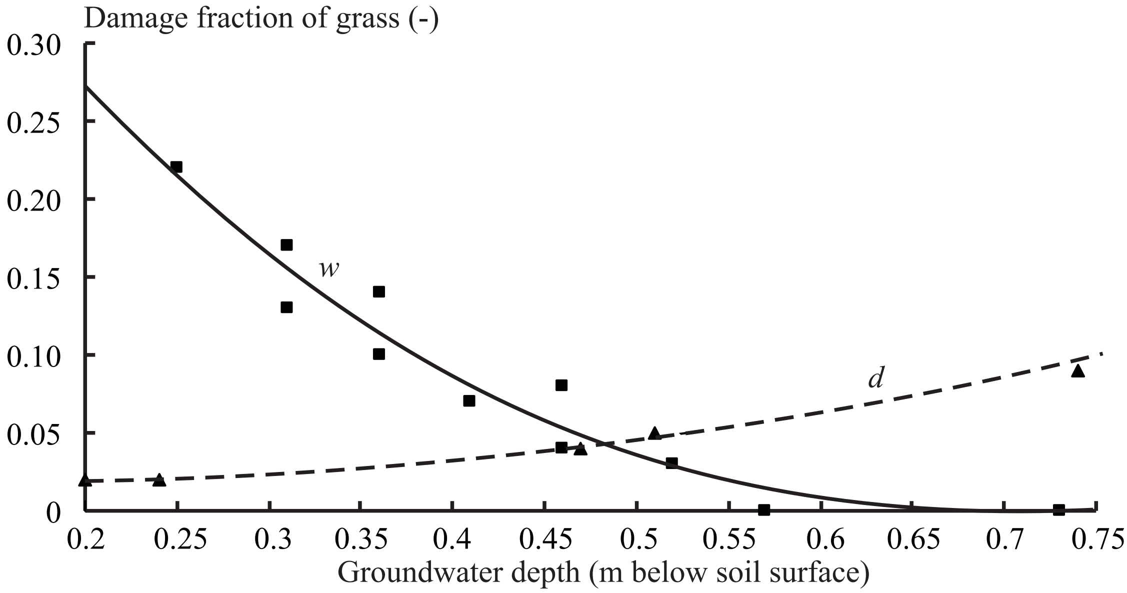

Figure 4.4 Damage fractions as a function of groundwater depth (the observations represent data from Working group HELP, 1987, code H1a, Table G7).

4.3 The allocation of compensation payments to characteristic nature reserves

The model is applied to analyse the cost-effectiveness of compensation payments for the restoration of desiccated nature reserves. An insight into the cost-effectiveness can be used to support decisions about the allocation of compensation payments among various nature reserves. The model is used to assess an increase in the average groundwater level in three hypothetical nature reserves in the eastern and southern part of the Netherlands, i.e. an oligotrophic bog *(m*=1*)*, a mesotrophic hay meadow *(m*=2*)* and an eutrophic hay meadow *(m*=3*)*. We assume that the present groundwater level is 0.3 m below the optimal groundwater level in each of these three reserves. The characteristics of these three nature reserves are derived from various kinds of maps, expert judgement and literature (e.g. Dijkema et al., 1985; Grootjans, 1985; Kemmers, 1986; van Buuren, 1997; van Diggelen, 1998; Spieksma, 1998; Jansen, 2000; Runhaar et al., 2000); see Appendix C (Table C1). The oligotrophic bog mainly consists of species-poor vegetation dominated by *Sphagnum* mosses and dwarf bushes (*Ericaceae*) on primarily peat. The mesotrophic hay meadow is relatively rich in species, with species like *Cirsium dissectum* and *Molinia coerulea*. The soil consists of sand. In the vegetation of the eutrophic hay meadow, species like *Caltha palustris* and *Filipendula ulmaria* are characteristic. The soil consists of peat. Estimation results of damage fractions as a function of groundwater depths are shown in Appendix C (Table C2).

The model is used to compare the allocation possibilities of a given budget for restoration. It can maximise environmental targets, i.e. nature restoration, in several ways: firstly, the total *increase in the unaffected surface areas* can be maximised (equation (4.7)); secondly, the total *increase in the ecological conservation values* can be maximised (equation (4.8)); thirdly, the total *utility of an increase in unaffected surface areas* can be maximised (equation (4.9)) by raising the increase in the unaffected surface area to the power of B_m, which reflects weights attached to restoration of reserve *m,* in a standard Cobb-Douglas utility function. Weights can be based on policy preferences or conservation criteria, e.g. rarity. The utility function, equation (4.9), is specified in such a way that the marginal utility of an additional hectare restored declines as more nature is restored, with the sum of weights equal to unity ΣB_m=1 (homogeneous Cobb-Douglas function specification).

$$\max \sum_{m=1}^{m=3} \Delta A_m^{\mathrm{u}} \qquad \text{subject to} \quad \sum_{m=1}^{m=3} \Delta C_m (\Delta A_m^{\mathrm{u}}) \leq Z \tag{4.7}$$

$$\max \sum_{m=1}^{m=3} \Delta V_m^{\mathrm{r}} \qquad \text{subject to} \quad \sum_{m=1}^{m=3} \Delta C_m (\Delta V_m^{\mathrm{r}}) \leq Z \tag{4.8}$$

$$\max \prod_{m=1}^{m=3} (\Delta A_m^{\mathrm{u}})^{B_m} \qquad \text{subject to} \quad \sum_{m=1}^{m=3} \Delta C_m (\Delta A_m^{\mathrm{u}}) \leq Z \tag{4.9}$$

These maximisation problems are subject to a number of restrictions. The increase in total agricultural costs due to higher groundwater levels ΔC is limited by the annual available amount of compensation payments for restoration, i.e. by a given budget Z (€/yr). These costs are a function of the increase in the unaffected surface area $\Delta C_m(\Delta A_m^{\mathrm{u}})$ or the increase in the conservation value $\Delta C_m(\Delta V_m^{\mathrm{r}})$ (see Figures 4.5 and 4.6). The increase in unaffected surface area is restricted by desiccated surface area in the base situation, while the increase in the conservation value is restricted by the loss in value in the base situation.

4.4 Results

In this section we provide insight into: (a) the cost-effectiveness of restoration in each of the three nature reserves; (b) the possible distribution of payments for restoration over the three nature reserves; and (c) the cost-effectiveness of investments in hydrological measures to re-wet nature.

a) Cost-effectiveness of restoration in each of the three nature reserves

The results of an increase in the average groundwater level of 0.3 m in the three reserves, calculated with the model, are shown in Table 4.5. Such an increase in the groundwater level can be achieved by an increase in the groundwater recharge rate (of 0.12-0.82 mm per day) or by creating a buffer zone on both sides of the reserve measuring ΔW (of 400-780 m). The three reserves are fully restored after the raise in the groundwater level.

The increase in the unaffected surface area is highest in the eutrophic hay meadow reserve (34.7 ha/yr) and lowest in the mesotrophic hay meadow reserve (15.3 ha/yr). However, the increase in the conservation value is highest in the mesotrophic hay meadow reserve (91 cvu/yr) and lowest in the eutrophic hay meadow reserve (20 cvu/yr). The increase in annual total agricultural costs ranges from €225/yr in the oligotrophic bog reserve to €1,004/yr in the eutrophic hay meadow reserve. When we combine results of the ecological and agricultural module, we get insight into the cost-effectiveness of the restoration of nature. Restoration is most effective (smallest increase in costs per unit increase in benefits) in the oligotrophic bog reserve. In the other two reserves *the cost-effectiveness depends on the indicator used.*

Table 4.5 Model results of an increase in the average groundwater level of 0.3 m in the three nature reserves.

	Oligotrophic bog	Mesotrophic hay meadow	Eutrophic hay meadow	Total
Increase in the groundwater recharge rate ΔR (m/d) or	0.00012	0.00082	0.00057	
Hydrological buffer zone ΔW (m)	780	580	400	
Increase in the unaffected surface area ΔA^{u} (ha/yr)	32.8	15.3	34.7	82.8
Increase in the conservation value ΔV^{r} (cvu/yr)	43	91	20	154
Increase in the wet damage costs ΔC^{w} (€/yr)	391	1508	1721	3620
Decrease in the drought damage costs ΔC^{d} (€/yr)	165	810	717	1692
Increase in the total agricultural costs ΔC (€/yr)	225	698	1004	1927
Cost-effectiveness indicator (equation 4.1) E^{A} (€/ha)	6.9	45.6	29.0	23.3
Cost-effectiveness indicator (equation 4.2) E^{v} (€/cvu)	5.2	7.6	50.4	12.5

The increase in the unaffected surface area and the increase in the conservation value, both as function of the increase in agricultural costs, are shown in Figures 4.5 and 4.6, respectively. Figure 4.5 shows that the restoration of desiccated surface area is more effective in the eutrophic hay meadow reserve than in the mesotrophic hay meadow reserve when more than 9 hectares are restored. Figure 4.6 shows, however, that the increase in the conservation value is least effective in the eutrophic reserve. The sensitivity of the results shown in Figure 4.6 to other conservation values ascribed to site types is shown in Appendix C (Figure C1).

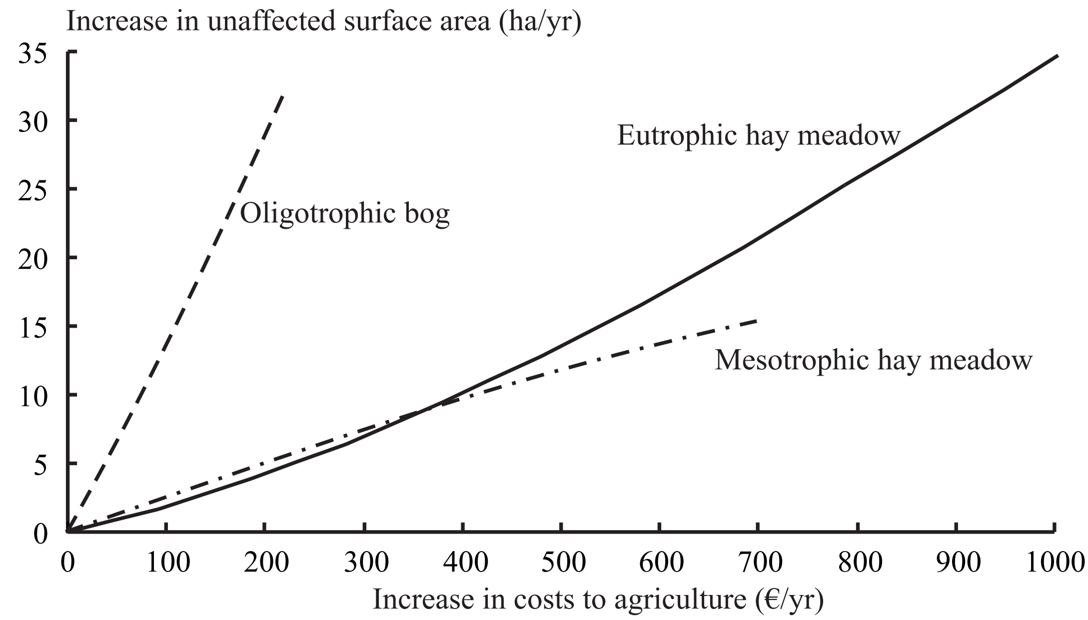

Figure 4.5 Increase in unaffected area as a function of the increase in agricultural costs in the three reserves.

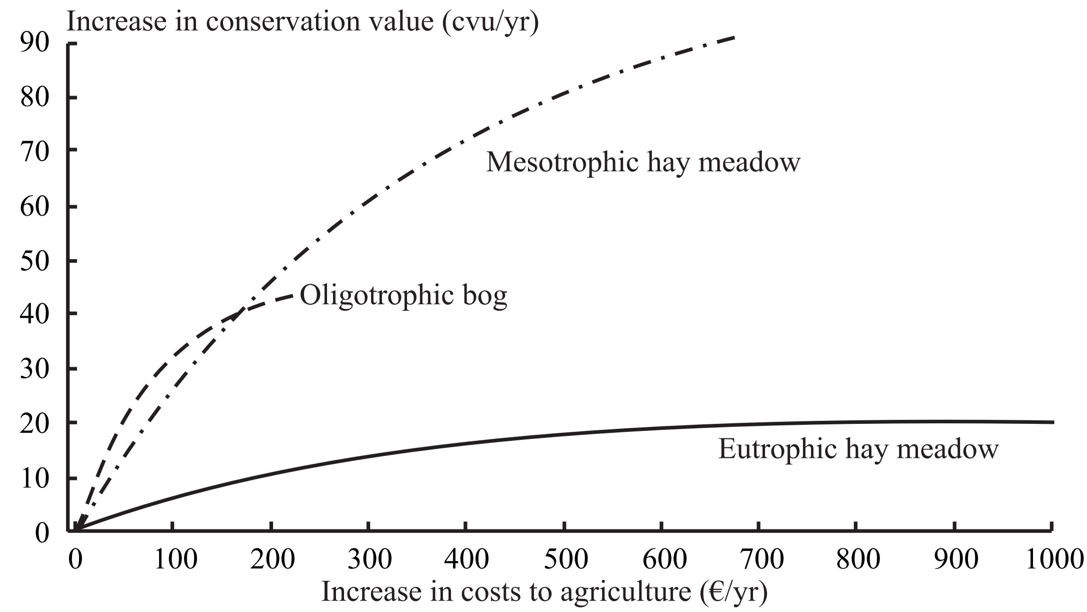

Figure 4.6 Increase in conservation value as a function of the increase in agricultural costs in the three reserves.

Estimation results of the increase in agricultural costs as a function of the increase in the unaffected surface area $\Delta C_m(\Delta A_m^{\mathrm{u}})$ and as a function of the increase in the conservation value $\Delta C_m(\Delta V_m^{\mathrm{r}})$ are shown in Appendix C (Table C3). Marginal cost functions of an increase in the unaffected surface area and increase in the conservation value can be derived from these functions by taking the first derivative with regard to ΔA_m^{u} and ΔV_m^{r}, see Figures 4.7 and 4.8.

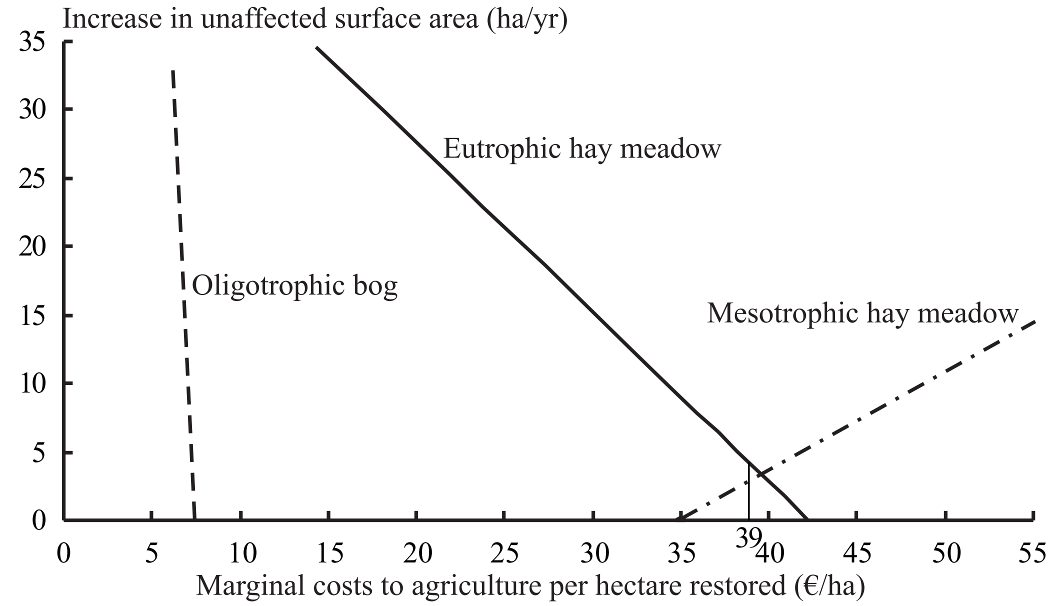

Figure 4.7 Marginal costs to agriculture of the restoration of the desiccated surface area in the three reserves.

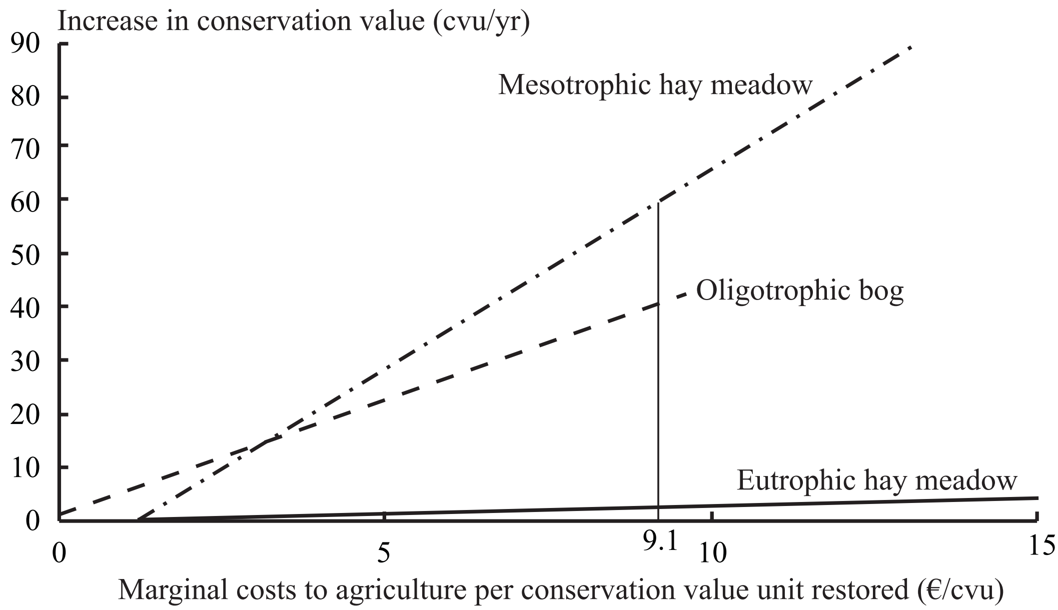

Figure 4.8 Marginal costs to agriculture of the restoration of the conservation value in the three reserves.

The marginal costs per conservation value unit restored increase as more nature is restored, whereas marginal costs per hectare restored may increase or decrease as more nature is restored (see Figures 4.8 and 4.7). Marginal costs increase in the mesotrophic hay meadow reserve, because the restoration rate is rather constant, which can be explained by the large standard deviation of the soil surface level (0.3 m). The marginal costs per hectare restored decrease in the other two reserves, because the restoration rate increases as more nature is restored due to the smaller standard deviation of the soil surface level (0.1 and 0.2 m).

b) Possible distribution of payments for restoration over the three nature reserves

The total annual agricultural costs of full (100%) restoration of the three reserves is €1,927/yr (see Table 4.5). The restoration percentages of the three reserves as a function of agricultural costs of restoration are presented in Figure 4.9 for the desiccated surface area and conservation value. If the total budget available to compensate farmers *Z* is €1,000/yr (which is about half the required budget for full restoration), more than 70% of the desiccated surface area (of 82.8 ha/yr) will be restored as will more than 90% of the conservation value losses (of 154 cvu/yr). With *Z* is €500/yr almost 50% of the desiccated surface area can be restored as can more than 65% of the conservation value losses.

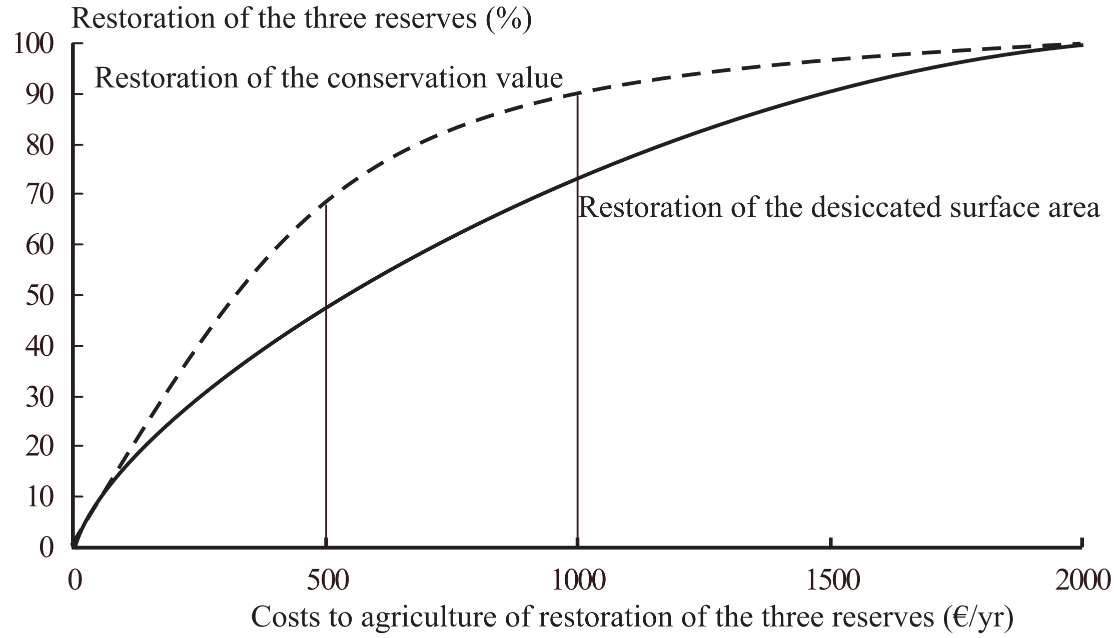

Figure 4.9 Restoration percentage as a function of costs to agriculture of restoration of the three reserves.

A given budget of *Z*= €500/yr can be allocated among reserves in different ways (see Table 4.6). If the total increase in the unaffected surface area is maximised (equation (4.7)), the marginal costs to agriculture per hectare restored will be €39/ha in both hay meadow reserves and €6.1/ha in the oligotrophic bog reserve, which will be fully restored (see Figure 4.7). If the total increase in the conservation value is maximised (equation (4.8)), marginal costs per conservation value unit restored will be €9.1/cvu in all reserves (Figure 4.8). Table 4.6 shows that only 3 ha of mesotrophic hay meadow reserve will be restored if we maximise the increase in the unaffected surface area, while 7.5 ha of mesotrophic hay meadow will be restored if we maximise the increase in the conservation value. If the total utility of an increase in unaffected surface area is maximised (equation (4.9)), substantial restoration will take place in all reserves because the marginal utility of restoration decreases in each reserve as more nature is restored, but less surface area will be restored in total.

Table 4.6 Maximisation of the increase in the unaffected surface area and increase in the conservation value and maximisation of the total utility of an increase in the unaffected surface area for several weights. Total refers to $\sum_{m=1}^{m=3} \Delta A_m^{\mathrm{u}}$, $\sum_{m=1}^{m=3} \Delta V_m^{\mathrm{r}}$, $\prod_{m=1}^{m=3} (\Delta A_m^{\mathrm{u}})^{B_m}$ and $\sum_{m=1}^{m=3} Z_m$.

Maximisation		Oligotrophic bog	Mesotrophic hay meadow	Eutrophic hay meadow	Total
$\sum_{m=1}^{m=3} \Delta A_m^{\mathrm{u}}$ equation (4.7)	ΔA^{u} (ha/yr)	32.8	3.0	4.1	**39.9**
	ΔV^{r} (cvu/yr)	43.2	27.7	8.7	79.6
	$(\Delta A^{\mathrm{u}})^{Bm}$ (-) with B_1=B_2=B_3=0.333	3.2	1.4	1.6	7.3
	Z (€/yr)	225	110	165	500
$\sum_{m=1}^{m=3} \Delta V_m^{\mathrm{r}}$ equation (4.8)	ΔA^{u} (ha/yr)	25.9	7.5	1.7	35.1
	ΔV^{r} (cvu/yr)	40.7	59.7	2.5	**102.9**
	$(\Delta A^{\mathrm{u}})^{Bm}$ (-) with B_1=B_2=B_3=0.333	3.0	2.0	1.2	6.9
	Z (€/yr)	181	306	13	500
$\prod_{m=1}^{m=3} (\Delta A_m^{\mathrm{u}})^{B_m}$ equation (4.9)	ΔA^{u} (ha/yr)	25.3	4.0	4.2	33.5
	ΔV^{r} (cvu/yr)	42.2	36.5	9.0	87.7
	$(\Delta A^{\mathrm{u}})^{Bm}$ (-) with B_1=B_2=B_3=0.333	2.9	1.6	1.6	**7.5**
	Z (€/yr)	177	152	171	500
$\prod_{m=1}^{m=3} (\Delta A_m^{\mathrm{u}})^{B_m}$ equation (4.9)	ΔA^{u} (ha/yr)	7.1	5.9	5.4	18.4
	ΔV^{r} (cvu/yr)	18.9	50.5	11.0	80.3
	$(\Delta A^{\mathrm{u}})^{Bm}$ (-) with B_1=0.1;B_2=0.5;B_3=0.4	1.2	2.4	2.0	**5.8**
	Z (€/yr)	52	231	216	500
$\prod_{m=1}^{m=3} (\Delta A_m^{\mathrm{u}})^{B_m}$ equation (4.9)	ΔA^{u} (ha/yr)	25.9	7.5	1.7	35.1
	ΔV^{r} (cvu/yr)	40.7	59.7	2.5	102.9
	$(\Delta A^{\mathrm{u}})^{Bm}$ (-) with B_1=0.29;B_2=0.63;B_3=0.08	2.6	3.6	1.0	**9.5**
	Z (€/yr)	181	306	13	500

Table 4.6 also shows that the allocation of payments, Z, among reserves is sensitive to the weights B_m, i.e. policy preferences. Even reserves where restoration is less effective will be restored if a large weight is attached to restoration. Weights can be chosen in such a way that restoration of the conservation value becomes equal to restoration under maximisation of the increase in the conservation value (see the results of the last maximisation in Table 4.6).

c) Cost-effectiveness of hydrological measures to re-wet nature

Cost-effectiveness analysis can also be used to compare the costs of hydrological measures to re-wet nature, which achieve the same goal. The kind of hydrological measures which can be used to re-wet nature, depends, however, on the nature reserve type. We will study here the investment costs of an increase in the groundwater recharge rate and the creation of a buffer zone, which both increase the groundwater level in the reserve by 0.3 m. Agricultural costs per hectare restored E^{A} (excluding investment costs) are equal for both hydrological measures (see Table 4.5).

The groundwater recharge rate can be increased at low costs by felling trees in order to reduce evaporation or by filling up superficial gullies that discharge water in wet periods. This is an interesting option mainly in some oligotrophic bog reserves, where the remnants of such gullies may be present as a result of agricultural practices in the past (before the twentieth century). The groundwater recharge rate can also be increased at low costs by the inlet of foreign water, by means of existing adjustable weirs or pumps. However, the inlet of foreign water is neither always technically possible nor ecologically desirable. It is not always technically possible because only a limited surface area of the sloping land can be supplied in the high part of the Netherlands. It is not always ecologically desirable since foreign water may lead to (internal) eutrophication. The inlet of foreign water is certainly ecologically undesirable in oligotrophic bog reserves, since they rely entirely on rainwater. The inlet of foreign water is an interesting option mainly in eutrophic hay meadow reserves.

If the felling of trees or the filling up of superficial gullies does not increase the groundwater recharge rate sufficiently in oligotrophic bog reserves, the reserve can in principle be separated from adjacent farmland by setting up partitions of foil in the soil. In mesotrophic hay meadow reserves, the most interesting solution is the creation of buffer zones between the reserve and the adjacent farmland. In eutrophic hay meadow reserves this will also be a promising option if the inlet of foreign water can not solve the entire problem.

Table 4.7 shows the investment costs of setting up partitions of foil in the soil in oligotrophic bog reserves. It also shows the interest costs of investments in buffer zones per hectare restored in mesotrophic and eutrophic hay meadow reserves. It appears that the investment cost of hydrological measures to re-wet nature are most expensive in mesotrophic hay meadow reserves. The way the figures in Table 4.7 are derived is explained below.

Table 4.7 Investment costs of partitions of foil in the oligotrophic bog reserve with a lifetime of 30 years and investment costs of buffer zones in both hay meadow reserves at an interest rate of 5%. The total cost-effectiveness indicators show the agricultural cost plus investment costs per ha restored or per cvu restored.

Costs	Oligotrophic bog	Mesotrophic hay meadow	Eutrophic hay meadow
Investment costs of partitions (€)	146,750		
Annual investment costs of partitions (€/yr)	4,892		
Investment costs of partitions per hectare restored (€ $ha^{-1}yr^{-1}$)	149		
Investment costs of partitions per cvu restored (€ $cvu^{-1}yr^{-1}$)	114		
Investment costs of buffer zones (€)		1740,000	1200,000
Annual interest costs of investment in buffer zones (€/yr)		87,000	60,000
Interest costs of buffer zones per hectare restored (€ $ha^{-1}yr^{-1}$)		5,686	1,729
Interest cost of buffer zones per cvu restored (€ $cvu^{-1}yr^{-1}$)		956	3,000
Total cost-effectiveness indicator (incl. investment cost) (€/ha)	156	5,732	1,758
Total cost-effectiveness indicator (incl. investment cost) (€/cvu)	119	964	3,050

Suppose the fixed cost of setting up partitions of foil in the soil is about €2,750 and the variable costs are €45 per metre circumference of reserves. Costs per hectare per year depend therefore on the size and shape of reserves. The setting up of partitions of foil in the soil is relatively cheap in large square or circular reserves, and expensive in small rectangular reserves. Since the circumference of the oligotrophic bog reserve is 3,200 metres, investment costs are €146,750. It is, however, only a temporary measure, since they have to be replaced after some years. If they have a lifetime of 30 years, annual costs will be €4,892/yr for 32.8 ha nature restored, which is €149 $ha^{-1}yr^{-1}$.

The investment costs of the creation of a buffer zone can be derived from the increase in the width of reserves (Table 4.5) times the length (Table C1) multiplied by the price of farmland, which was about €30,000/ha in the Netherlands in 1999 (Silvis and van Bruchem, 2000). Investment costs in the mesotrophic and eutrophic hay meadow reserves are €1.74 million and €1.2 million, since 58 ha and 40 ha have to be purchased for the creation of buffer zones. As deprecation of land is zero, only the annual interest costs have to be taken into account, which is €87,000/yr and €60,000/yr at a rate of interest of 5%. This is €5,686 $ha^{-1}yr^{-1}$ and €1,729 $ha^{-1}yr^{-1}$ respectively (Table 4.7). Since buffer zones can be used for all kinds of purposes, the net costs are smaller. Buffer zones can be added to nature reserves or used to cultivate wet-tolerant crops, like the energy crop willow (Londo et al., 2000). Net costs thus depend on these benefits, which determine the cost-effectiveness of measures.

The total cost-effectiveness of restoration presented in Table 4.7 consists of an increase in total agricultural cost (see Table 4.5) plus the investment costs of hydrological measures. Table 4.7 shows that restoration is most cost-efficient in oligotrophic bog reserves. It is least cost-efficient in terms of an increase in the unaffected surface area in the mesotrophic hay meadow reserve. In terms of an increase in the conservation value it is least cost-efficient in the eutrophic hay meadow reserve.

4.5 Discussion and conclusions

The model makes it possible to compare differences in the cost-effectiveness of an increase in the ecological conservation value of desiccated nature. It is, however, important to note that in two approaches we ascribe constant marginal values to site types, whereas these marginal values vary with the national rarity of site types. For this reason a specification with a declining marginal value of nature as in the utility function approach is more appropriate. The impact of neglecting such a change in the value of site types on the results will be small in this study, since only three small nature reserves are restored and the impact on the total size of site types at the national level seems to be modest. Besides, ecological restoration is a long-term process. Moreover, conservation values ascribed to site types in this study are only reference values and the results are sensitive to other values, as shown in Appendix C (Figure C1). Users of the model may ascribe their own values to site types.

It is also important to note that the model has limitations, due to various simplifications. Firstly, in the ecological module the quality of groundwater is not taken into account. This quality is especially important in areas with upward seepage. Secondly, the model has limitations with respect to the shape of the nature reserves, which can not always be modelled as an elongated reserve. It is therefore recommended to further develop the model.

Second-order effects are also not considered in the analysis. Changes in the location of crops and in cropping patterns for instance have not been assessed, because the impact seems to be modest because of limited adjustment possibilities. Almost all (95%) of adjacent farmland consists in our case study areas of grass, which is more wet-tolerant than arable crops.

In this study a prototype multidisciplinary model has been developed, which provides insight into the cost-effectiveness of restoration projects that reduce the desiccation of nature. Despite its simplicity, the model is useful to support decisions about the allocation of compensation payments to restoration projects. Different objective functions, reflecting policy targets, can be formulated. The model can be used to maximise: (1) the total increase in the unaffected surface area; (2) the total increase in the ecological conservation value; and (3) the total utility of an increase in unaffected surface area. The cost-effectiveness appears to be dependent on the kind of indicator used, while the second indicator is sensitive to conservation values ascribed to site types. The utility approach is sensitive to the weights, but has the advantage of decreasing marginal utility.

The study shows that the current policy goal to reduce the desiccated surface area will not maximise the increase in the ecological conservation value. The allocation of a given budget can be improved in terms of ecological restoration. Since marginal costs to agriculture per conservation value unit restored increase as more nature is restored, the partial restoration of many reserves rather than the full restoration of a few reserves could be considered.

The study also shows that agricultural costs are small compared to the investment costs of hydrological measures to re-wet nature, which vary considerably. The groundwater recharge rate can be increased at low costs, but this is neither always technically possible nor ecologically desirable. Unambiguous statements about the size of net investment costs of the creation of buffer zones are hard to make without an insight into ancillary benefits from the use of these zones (in additional to the benefits of restoration). Nevertheless, it is clear that ancillary benefits must be substantial in order to cover the costs of creating buffer zones.

Appendix A
Hydrological module of an elongated nature reserve

In this appendix we will first determine the hydraulic head in the aquifer beneath the nature reserve and the farmland, as well as the average groundwater level in the nature reserve relative to the surface water level (A.1). Then we will describe how our module computes the groundwater level in farmland relative to the surface water level (A.2). In (A.3) we explain how the groundwater level and soil surface levels are used to calculate the depth of the groundwater table beneath soil surface. Horizontal distances *X* are relative to the middle of the reserve and the reference of both hydraulic head and soil surface is the surface water level of farmland. We will end with a description of re-wetting scenarios (A.4).

A.1 Hydraulic head in the aquifer and average groundwater level in the nature reserve

For the calculation of the hydraulic head in the aquifer H^1 as a function of place *X*, we use the scheme of Figure A1.

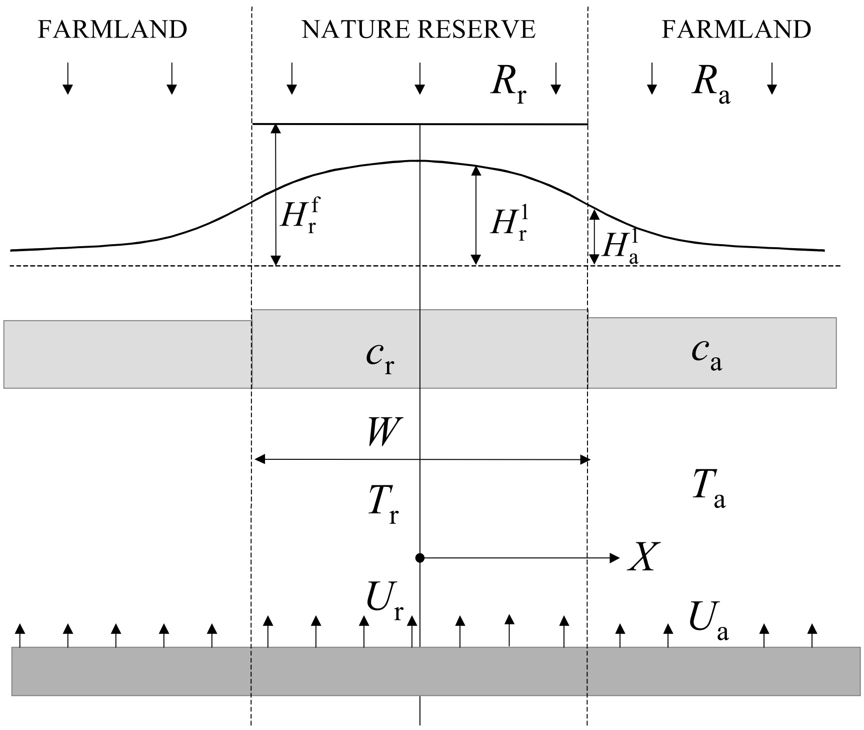

Figure A1. Schematisation of the elongated nature reserve and the adjacent farmland.

An elongated nature reserve of width W is neighboured on both sides by farmland with a controlled surface water level. As a result of a stationary groundwater recharge rate R and a regional upward seepage rate U, there is a difference between the average groundwater level in the nature reserve and the surface water level of the farmland and groundwater flows in the direction of the farmland. Farmland and nature reserves differ with respect to geo-hydrological properties: the hydraulic resistance of the top layer c, the transmissivity of the underlying aquifer T, and the intensity of the regional upward seepage rate U. We used the Dupuit-Forcheimer assumption and suggestions by Bruggeman (1999, p. 148, 631) to solve this problem.

The differential equation for groundwater flow in the aquifer below the nature reserve is:

$$\frac{\mathrm{d}^2 H_\mathrm{r}^1}{\mathrm{d}X^2} - \frac{H_\mathrm{r}^1}{\lambda_\mathrm{r}^2} + \frac{U_\mathrm{r}}{T_\mathrm{r}} = 0 \qquad \text{(A.1)}$$

where:

H_r^1 = hydraulic head aquifer below nature reserve [L]

λ_r = leakage factor nature reserve [L]: $\lambda_\mathrm{r} = \sqrt{T_\mathrm{r} c_\mathrm{r}}$

T_r = transmissivity aquifer nature reserve [L^2T^{-1}]

c_r = hydraulic resistance top-layer nature reserve [T]

U_r = regional upward seepage rate under nature reserve [$L.T^{-1}$]

X = distance to the middle of the nature reserve [L]

The differential equation for groundwater flow in the aquifer below the farmland is:

$$\frac{\mathrm{d}^2 H_\mathrm{a}^1}{\mathrm{d}X^2} - \frac{H_\mathrm{a}^1}{\lambda_\mathrm{r}^2} + \frac{U_\mathrm{a}}{T_\mathrm{a}} = 0 \qquad \text{(A.2)}$$

where:

H_a^1 = hydraulic head aquifer below farmland [L]

λ_a = leakage factor farmland [L]: $\lambda_\mathrm{a} = \sqrt{T_\mathrm{a} c_\mathrm{a}}$

T_a = transmissivity aquifer farmland [L^2T^{-1}]

c_a = hydraulic resistance top-layer farmland [T]

U_a = regional upward seepage rate under farmland [$L.T^{-1}$]

Furthermore, this problem has the following five conditions:

$$\frac{\mathrm{d}H_{\mathrm{r}}^{1}}{\mathrm{d}X}(0)=0 \tag{A.3}$$

$$\frac{\mathrm{d}H_{\mathrm{a}}^{1}}{\mathrm{d}X}(\infty)=0 \tag{A.4}$$

$$H_{\mathrm{r}}^{1}\left(\tfrac{1}{2}W\right)=H_{\mathrm{a}}^{1}\left(\tfrac{1}{2}W\right) \tag{A.5}$$

$$\frac{\mathrm{d}H_{\mathrm{r}}^{1}}{\mathrm{d}X}\left(\tfrac{1}{2}W\right)=\frac{\mathrm{d}H_{\mathrm{a}}^{1}}{\mathrm{d}X}\left(\tfrac{1}{2}W\right) \tag{A.6}$$

$$\int_{X=0}^{X=\frac{1}{2}W}\frac{H_{\mathrm{r}}^{\mathrm{f}}-H_{\mathrm{r}}^{1}}{c_{\mathrm{r}}}=\tfrac{1}{2}R_{\mathrm{r}}W \tag{A.7}$$

where:

$H_{\mathrm{r}}^{\mathrm{f}}$ = groundwater level in the nature reserve relative to the surface water level [L]

R_{r} = groundwater recharge rate nature reserve [L.T^{-1}]

The solution of (A.1)-(A.7) is (for suggestions, see Bruggeman (1999, p. 148, 631)):

$$H_{\mathrm{r}}^{\mathrm{f}}=U_{\mathrm{a}}c_{\mathrm{a}}-U_{\mathrm{r}}c_{\mathrm{r}}+\frac{Wc_{\mathrm{r}}\left(U_{\mathrm{r}}+R_{\mathrm{r}}\right)}{2\lambda_{\mathrm{r}}^{2}}\left(\lambda_{\mathrm{a}}+\lambda_{\mathrm{r}}\tanh^{-1}\left(\frac{W}{2\lambda_{\mathrm{r}}}\right)\right)\text{ and} \tag{A.8}$$

and:

$$H_{\mathrm{a}}^{1}(X)=U_{\mathrm{a}}c_{\mathrm{a}}+\frac{\lambda_{\mathrm{a}}Wc_{\mathrm{r}}\left(U_{\mathrm{r}}+R_{\mathrm{r}}\right)}{2\lambda_{\mathrm{r}}^{2}}\exp\left(\frac{W-2X}{2\lambda_{\mathrm{a}}}\right) \tag{A.9}$$

A.2 Groundwater level in farmland relative to soil surface

We assume that the resistance of the top-layer in the farmland is the sum of two resistances: a vertical resistance for the flow of water from the aquifer to the parcel, and a drainage resistance, for the flow from the parcel to its adjacent ditches:

$$c_{\mathrm{a}}=c_{\mathrm{a}}^{\mathrm{vert}}+c_{\mathrm{a}}^{\mathrm{dran}} \tag{A.10}$$

where:

$c_{\mathrm{a}}^{\mathrm{vert}}$ = vertical resistance farmland [T]

$c_{\mathrm{a}}^{\mathrm{dran}}$ = drainage resistance farmland [T]

Using (A.10), we can now compute the groundwater level in farmland relative to the surface water level as (equation 17 in van Drecht, 1997):

$$H_{\mathrm{a}}^{\mathrm{f}} = \frac{c_{\mathrm{a}}^{\mathrm{dran}} c_{\mathrm{a}}^{\mathrm{vert}} R_{\mathrm{a}} + c_{\mathrm{a}}^{\mathrm{dran}} H_{\mathrm{a}}^{1}}{c_{\mathrm{a}}^{\mathrm{vert}} + c_{\mathrm{a}}^{\mathrm{dran}}} \tag{A.11}$$

where:

$H_{\mathrm{a}}^{\mathrm{f}}$ = groundwater level in farmland relative to the surface water level [L]

R_{a} = groundwater recharge rate farmland [L.T^{-1}]

If the upper resistance layer consists of phreatic aquifer on top of an aquitard, the vertical resistance is $c_{\mathrm{a}}^{\mathrm{vert}} = c_1 + \frac{M}{k_{\mathrm{vert}}}$. With $k_{\mathrm{vert}} = 0.2 k_{\mathrm{hor}}$ and $k_{\mathrm{hor}} = \frac{T_{\mathrm{a}}}{M}$ we get:

$$c_{\mathrm{a}}^{\mathrm{vert}} = c_1 + \frac{M^2}{0.2 T_{\mathrm{a}}} \tag{A.12}$$

where:

c_1 = vertical resistance aquitard [T]

M = distance between the top of the aquitard and the ditch water level [L]

k_{vert} = vertical hydraulic conductivity phreatic aquifer [L.T^{-1}]

k_{hor} = horizontal hydraulic conductivity phreatic aquifer [L.T^{-1}]

A.3 Groundwater depth

We assume that the soil surface of the farmland is flat. Using this assumption, the groundwater depth of the farmland is calculated as:

$$D_{\mathrm{a}} = S_{\mathrm{a}}^{\mathrm{s}} - H_{\mathrm{a}}^{\mathrm{f}} \tag{A.13}$$

where:

D_{a} = groundwater depth of the farmland (relative to soil surface) [L]

$S_{\mathrm{a}}^{\mathrm{s}}$ = average soil surface level of the farmland [L]

In contrast to farmland, nature reserves are usually gently sloping. To account for this, we assume the soil surface of the reserve to be normally distributed:

$$D_\mathrm{r} = S_\mathrm{r}^\mathrm{s} - H_\mathrm{r}^\mathrm{f} \tag{A.14}$$

$$S_\mathrm{r}^\mathrm{s} \approx \mathrm{N}(\upsilon, \sigma^2) \tag{A.15}$$

where:

D_r = groundwater depth of the nature reserve (relative to soil surface) [L]

S_r^s = soil surface level of the nature reserve [L]

υ = average soil surface level of the nature reserve [L]

σ = standard deviation of the soil surface level of the nature reserve [L]

When we combine equation (A.14) and equation (A.15), it turns out that the groundwater depth in the nature reserve is normally distributed as well.

$$D_\mathrm{r} \approx \mathrm{N}(\upsilon - H_\mathrm{r}^\mathrm{f}, \sigma^2) \tag{A.16}$$

A.4 Re-wetting scenarios

There are numerous ways to re-wet a nature reserve. We will examine only two extreme ways: a very cheap one and a very expensive one.

One can try to enlarge the groundwater recharge rate in the nature reserve, R_r, for instance by felling trees in order to reduce evaporation, by filling up superficial gullies or by the inlet of surface water. Suppose we want to raise the average groundwater level in the nature reserve by $\Delta H_\mathrm{r}^\mathrm{f}$. From (A.8) it follows that this requires an additional recharge of:

$$\Delta R_\mathrm{r}\left(\Delta H_\mathrm{r}^\mathrm{f}\right) = \frac{2\lambda_\mathrm{r}^2 \Delta H_\mathrm{r}^\mathrm{f}}{W c_\mathrm{r}\left(\lambda_\mathrm{a} + \lambda_\mathrm{r} \tanh^{-1}\left(\dfrac{W}{2\lambda_\mathrm{r}}\right)\right)} \tag{A.17}$$

For both technical and ecological reasons it is often neither possible nor desirable to enlarge R_a. Another option is to create a zone with a relatively high groundwater level around the nature reserve. A simple albeit rough way to calculate the size of such a buffer zone is to enlarge the area that has a high water level by ΔW. From (A.8) it follows that, if we want to raise the average groundwater level in the nature reserve by $\Delta H_\mathrm{r}^\mathrm{f}$, this requires an extra width of:

$$\Delta W\left(\Delta H_\mathrm{r}^\mathrm{f}\right) = \frac{2\lambda_\mathrm{r}^2 \Delta H_\mathrm{r}^\mathrm{f}}{c_\mathrm{r}\left(U_\mathrm{r} + R_\mathrm{r}\right)}\left(\lambda_\mathrm{a} + \lambda_\mathrm{r} \tanh^{-1}\left(\frac{W}{2\lambda_\mathrm{r}}\right) - \tfrac{1}{2} W \operatorname{csch}\left(\frac{W^2}{4\lambda_\mathrm{r}^2}\right)\right)^{-1} \tag{A.18}$$

Appendix B
Agricultural module

Changes in the groundwater level can have a direct and an indirect effect on the crop. When the groundwater level reaches the root zone of the plants, it directly limits the availability of oxygen. It generally also results in a decrease in the workability of the soil and thus in a delay of planting and emergence date, which reduces yields (Feddes and van Wijk, 1990). At lower levels, the groundwater level has an indirect effect on crop production, because it determines through capillary rise the availability of moisture in root zones.

In this appendix we will first discuss the steps taken to relate groundwater depth to the wet- and drought-damage fractions (B.1). Then we will show how changes in wet- and drought-damage fractions of potential crop yields can be derived (B.2).

B.1 Relationship between the wet and drought-damage fractions and groundwater depth

The wet- and drought-damage fractions of potential physical yields (w and d) are derived from the yield losses tables developed by Working group HELP (1987), which are described in more detail in Section 3.3.

The yield losses tables developed by Working group HELP (1987) relate wet-damage fractions to the highest groundwater depth D^{h} and drought-damage fractions to the lowest groundwater depth D^{l}. Two steps are taken to relate these damage fractions to the groundwater depth D. Firstly, D is derived from the D^{h} and D^{l} on the basis of a formula developed by van der Sluijs (1982) (equation (B.1)). Secondly the relationships between D and D^{h} and between D and D^{l} (equations (B.2) and (B.3)) are estimated.

$$D = 0.054 + 0.83D^{\mathrm{h}} + 0.19D^{\mathrm{l}} \qquad \text{(B.1)}$$

$$D = 0.20 + 1.06D^{\mathrm{h}} \qquad \text{(Adjusted } R^2 \text{ is 0.99)} \qquad \text{(B.2)}$$

$$D = -0.35 + 0.78D^{\mathrm{l}} \qquad \text{(Adjusted } R^2 \text{ is 0.90)} \qquad \text{(B.3)}$$

As the yield losses tables are discrete and a continuous relationship between the damage fractions and groundwater depth is desirable, this relationship is estimated for the cubic functional form. The ordinary least squares method is used.

$$w = \beta_0^{\mathrm{w}} + \beta_1^{\mathrm{w}} D + \beta_2^{\mathrm{w}} D^2 + \beta_3^{\mathrm{w}} D^3 \tag{B.4}$$

$$d = \beta_0^{\mathrm{d}} + \beta_1^{\mathrm{d}} D + \beta_2^{\mathrm{d}} D^2 + \beta_3^{\mathrm{d}} D^3 \tag{B.5}$$

Changes in the wet- and drought-damage fractions of potential crop yields Δw and Δd, which are needed to obtain C^{w} and C^{d}, are equal to fractions after the rise in the groundwater level, minus fractions before the rise in the groundwater level (indicated with the subscripts ‘2’ and ‘1’ respectively).

$$\Delta w = w_2 - w_1 = \beta_1^{\mathrm{w}}(D_2 - D_1) + \beta_2^{\mathrm{w}}(D_2^2 - D_1^2) + \beta_3^{\mathrm{w}}(D_2^3 - D_1^3) \tag{B.6}$$

$$\Delta d = d_2 - d_1 = \beta_1^{\mathrm{d}}(D_2 - D_1) + \beta_2^{\mathrm{d}}(D_2^2 - D_1^2) + \beta_3^{\mathrm{d}}(D_2^3 - D_1^3) \tag{B.7}$$

B.2 Changes in wet- and drought-damage fractions

From (A.9), (A.11) and (A.13) it follows that:

$$D_1 = \vartheta - \nu_1 \exp(x^*) \tag{B.8}$$

$$D_2 = \vartheta - \nu_2 \exp(x^*) \tag{B.9}$$

with

$$\vartheta = S_{\mathrm{a}} - \frac{c_{\mathrm{a}}^{\mathrm{dran}} c_{\mathrm{a}}^{\mathrm{vert}} R_{\mathrm{a}}}{c_{\mathrm{a}}} - U_{\mathrm{a}} c_{\mathrm{a}}^{\mathrm{dran}}$$

and

$$x^* = \frac{W - 2X}{2\lambda_{\mathrm{a}}} \qquad \left(\mathrm{d}x^* = \mathrm{d}\frac{W}{2\lambda_{\mathrm{a}}} - \mathrm{d}\frac{X}{\lambda_{\mathrm{a}}} \Rightarrow \mathrm{d}X = -\lambda_{\mathrm{a}} \mathrm{d}x^*\right)$$

In the case of an increase in R_{r} from R_{r1} to R_{r2}:

$$\nu_1 = \frac{c_{\mathrm{a}}^{\mathrm{dran}} \lambda_{\mathrm{a}} W c_{\mathrm{r}} (U_{\mathrm{r}} + R_{\mathrm{r1}})}{2 c_{\mathrm{a}} \lambda_{\mathrm{r}}^2} \text{ and } \nu_2 = \frac{c_{\mathrm{a}}^{\mathrm{dran}} \lambda_{\mathrm{a}} W c_{\mathrm{r}} (U_{\mathrm{r}} + R_{\mathrm{r2}})}{2 c_{\mathrm{a}} \lambda_{\mathrm{r}}^2}$$

In the case of an increase in W from W_1 to W_2:

$$\nu_1 = \frac{c_{\mathrm{a}}^{\mathrm{dran}} \lambda_{\mathrm{a}} W_1 c_{\mathrm{r}} (U_{\mathrm{r}} + R_{\mathrm{r}})}{2 c_{\mathrm{a}} \lambda_{\mathrm{r}}^2} \text{ and } \nu_2 = \frac{c_{\mathrm{a}}^{\mathrm{dran}} \lambda_{\mathrm{a}} W_2 c_{\mathrm{r}} (U_{\mathrm{r}} + R_{\mathrm{r}})}{2 c_{\mathrm{a}} \lambda_{\mathrm{r}}^2}$$

To obtain C^{w} and C^{d} (equations (4.5) and (4.6)) we have to integrate the changes in wet- and drought-damage fractions over the distance X of farmland relative to the middle of the nature reserve. We first substitute equations (B.8) and (B.9) into equations (B.6) and (B.7), which gives the following equation (ignoring the superscripts 'w' and 'd'):

$$C^{\mathrm{w}} = \int_{X=\frac{W}{2}}^{X=\infty} \Delta w(X)\, \mathrm{d}X = -\lambda_{\mathrm{a}} \int_{x^*=0}^{x^*=-\infty} \Delta w(x^*)\, \mathrm{d}x^* =$$

$$-\lambda_{\mathrm{a}} \int_{x^*=0}^{x^*=-\infty} \begin{bmatrix} \beta_1^{\mathrm{w}} (\nu_1 - \nu_2) \exp(x^*) + \beta_2^{\mathrm{w}} (2\vartheta(\nu_1 - \nu_2) \exp(x^*) + (\nu_2^2 - \nu_1^2) \exp(2x^*)) \\ + \beta_3^{\mathrm{w}} (3\vartheta^2 (\nu_1 - \nu_2) \exp(x^*) + 3\vartheta(\nu_2^2 - \nu_1^2) \exp(2x^*) + (\nu_1^3 - \nu_2^3) \exp(3x^*)) \end{bmatrix} \mathrm{d}x^* \quad \text{(B.10)}$$

Integration gives the following result:

$$C^{\mathrm{w}} = \lambda_{\mathrm{a}} \begin{pmatrix} -(\beta_1^{\mathrm{w}} + \vartheta(2\beta_2^{\mathrm{w}} + 3\beta_3^{\mathrm{w}} \vartheta))(\nu_2 - \nu_1) \\ + 0.5(\beta_2^{\mathrm{w}} + 3\beta_3^{\mathrm{w}} \vartheta)(\nu_2^2 - \nu_1^2) - 0.33\beta_3(\nu_2^3 - \nu_1^3) \end{pmatrix} \quad \text{(B.11)}$$

C^{d} is derived in a similar was as C^{w}:

$$C^{\mathrm{d}} = \lambda_{\mathrm{a}} \begin{pmatrix} -(\beta_1^{\mathrm{d}} + \vartheta(2\beta_2^{\mathrm{d}} + 3\beta_3^{\mathrm{d}} \vartheta))(\nu_2 - \nu_1) \\ + 0.5(\beta_2^{\mathrm{d}} + 3\beta_3^{\mathrm{d}} \vartheta)(\nu_2^2 - \nu_1^2) - 0.33\beta_3(\nu_2^3 - \nu_1^3) \end{pmatrix} \quad \text{(B.12)}$$

Appendix C
Input, estimation results and sensitivity analysis

Table C1. Input needed for the hydrological, ecological and agricultural module. Inputs are based on Dijkema et al. (1985), Grootjans (1985), Kemmers (1986), van Buuren (1997), van Diggelen (1998), Spieksma (1998), Jansen (2000) and Runhaar et al. (2000).

	Oligotrophic bog		Mesotrophic hay meadow		Eutrophic hay meadow	
	Farm-land	Reserve	Farm-land	Reserve	Farm-land	Reserve
Groundwater recharge rate R (m/d)	0.0007	0.0007	0.0007	0.0007	0.0007	0.0007
Transmissivity of the aquifer T (m^2/d)	500	500	500	500	500	500
Upward seepage rate U (m/d)	0	0	0.0005	0.0005	0.001	0.001
Hydraulic resistance top-layer c_r (d) [a)]		2300		250		400
Drainage resistance c^{dran} (d)	100		100		100	
Vertical resistance aquitard c_1 (d)	0		0		20	
Distance top aquitard and ditch M (m)	20		20		20	
Average soil surface level μ (m)	0.55	2.05	0.65	0.65	0.7	0.9
Standard deviation soil surface level σ (m)		0.2		0.3		0.1
Average groundwater level H^f (m)		1.75		0.37		0.62
Width of the nature reserve W (m)		600		400		400
Length of the nature reserve L (m)		1000		1000		1000
Unaffected surface area A^u (ha)		27.2		24.7		5.3
Conservation value V^r (cvu/yr)		752		718		127
Ecological soil type number [b)]		102(0)		312(2)		104(3)
Fraction of grass in total area F (-) [c)]	0.95		0.95		0.95	
Fraction of maize in total area F (-) [c)]	0.05		0.05		0.05	
Physical yield of grass Y (tonne/ha) [d)]	61.3		61.3		61.3	
Physical yield of maize Y (tonne/ha) [d)]	43.3		43.3		43.3	
Price of maize P (€/tonne)	38.7		38.7		38.7	
Price of grass P (€/tonne)	21.2		21.2		21.2	

a) By changing c_r, we calibrate the model in such a way that H_r^f (equation (A.8)) becomes 0.3 m below the optimal level of H_r^f in each of these three reserves.

b) Occurrence probabilities correspond with these soil type numbers; see Klijn et al. (1997), Appendix 4.

c) We assume that the cropping pattern, physical crop yields and crop prices are equal in the three reserves.

d) These yields are supposed to be optimal yields, since groundwater levels are at present attuned to agriculture.

Table C2. Estimation results of fractions of wet- and drought-damage as a function of groundwater depth; equations (B.4) and (B.5). The wet- and drought-damage fractions for oligotrophic bog and mesotrophic hay meadow are derived from (code H1a, number 58 of Tables G7 and B7) for sandy podzol soil. The wet- and drought-damage fractions for eutrophic hay meadow are derived from (code aV, number 2 of Tables G1 and B1) for peaty soil (Working group HELP, 1987).

	$w=\beta_0^{\mathrm{w}}+\beta_1^{\mathrm{w}}D+\beta_2^{\mathrm{w}}D^2+\beta_3^{\mathrm{w}}D^3$					$d=\beta_0^{\mathrm{d}}+\beta_1^{\mathrm{d}}D+\beta_2^{\mathrm{d}}D^2+\beta_3^{\mathrm{d}}D^3$				
	β_0^{w}	β_1^{w}	β_2^{w}	β_2^{w}	Adjusted R^2	β_0^{d}	β_1^{d}	β_2^{d}	β_3^{d}	Adjusted R^2
Oligotrophic bog and mesotrophic hay meadow										
grass	0.61	-2.11	2.33	-0.81	0.93	0.04	-0.22	0.54	-0.19	0.99
maize	0.73	-1.82	1.45	-0.36	0.97	0.07	-0.29	0.6	-0.21	0.99
Eutrophic hay meadow										
grass	0.68	-1.83	1.38	-0.17	0.93	0.04	-0.09	0.39	-0.11	0.99
maize	0.83	-1.90	1.45	-0.35	0.93	-0.10	0.34	-0.01		0.99

Table C3. Estimation results of the increase in the agricultural costs of restoration as a function of the increase in the unaffected surface area and as a function of the increase in the ecological conservation value.

	$\Delta C_m=\alpha_1^{\mathrm{A}}\Delta A_m^{\mathrm{u}}+\alpha_2^{\mathrm{A}}(\Delta A_m^{\mathrm{u}})^2$			$\Delta C_m=\alpha_1^{\mathrm{V}}\Delta V_m^{\mathrm{r}}+\alpha_2^{\mathrm{V}}(\Delta V_m^{\mathrm{r}})^2$		
	α_1^{A}	α_2^{A}	Adjusted R^2	α_1^{V}	α_2^{V}	Adjusted R^2
Oligotrophic bog	7.54	-0.02	1.00	-0.24	0.11	0.98
Mesotrophic hay meadow	34.83	0.70	1.00	1.15	0.07	0.99
Eutrophic hay meadow	42.26	-0.40	0.99	1.33	1.57	0.97

Sensitivity analysis

Figure C1 provides insight into the sensitivity of results to other conservation values ascribed to site types. It shows three curves for each nature reserve (shown by the same kind of line). The middle curve for each nature reserve is based on the conservation values shown in Table 4.2 and used in this chapter (see Figure 4.6). The upper curve is based on higher conservation values deduced from Table 4.2 by squaring original values. The lower curve is based on lower conservation values deduced from Table 4.2 by taking the square root of the original values.

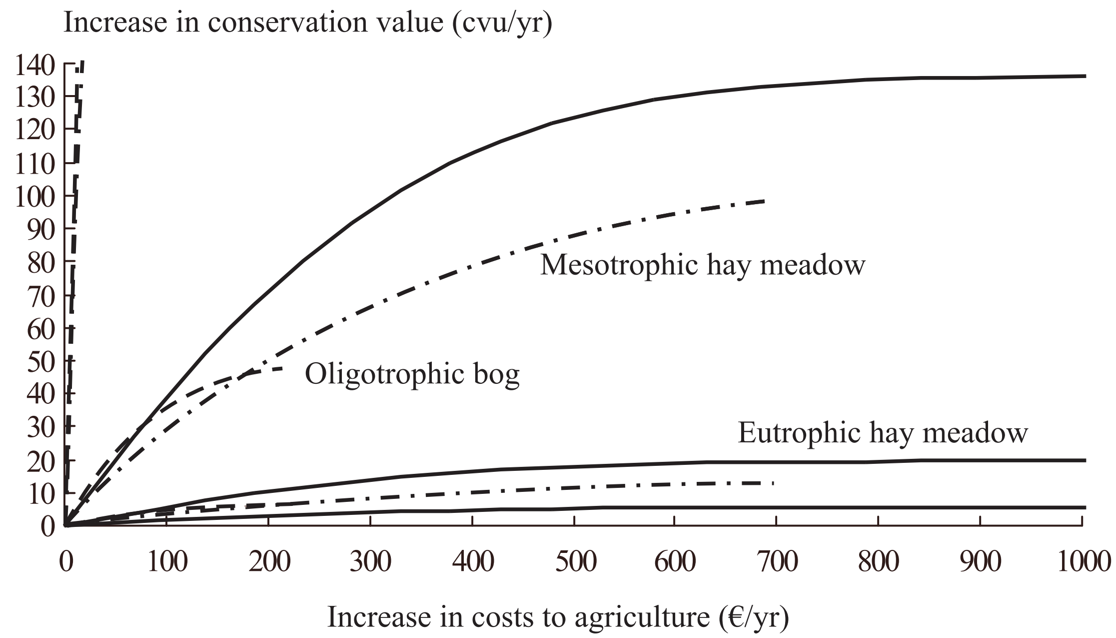

Figure C1. Sensitivity of the cost-effectiveness of an increase in the conservation value to other conservation values of site types in the three nature reserves.

The upper curves of the mesotrophic hay meadow reserve and oligotrophic bog reserve are only partially shown in Figure C1, since their values increase towards 2816 cvu/yr and 1082 cvu/yr respectively. Figure C1 shows that the relative difference in the cost-effectiveness of an increase in the conservation value among nature reserves is sensitive to the values ascribed to site types.

5 Groundwater extraction management[6]

5.1 Introduction

In the Netherlands, farmers extract shallow groundwater of a high quality for low-value use, e.g. in particular sprinkling. This extraction can result in the desiccation of adjacent nature reserves due to declining groundwater levels and the degradation of the quality of groundwater. Often these externalities are not considered when groundwater extraction paths are being determined.

Despite the seriousness of the pollution and desiccation problem in the Netherlands, economic literature on the internalisation of externalities from agricultural groundwater extraction is limited. Two well-developed branches of economic literature focus on groundwater. One focuses on water quantity, and emphasises the comparison between optimal extraction paths and common property outcomes (e.g. Gisser and Sanchez, 1980; Provencher and Burt, 1994). The other focuses on water quality and analyses contamination in a pollution-control perspective, with special emphasis on non-point pollution as an externality imposed by agricultural production activities (e.g. Larson et al., 1996; Fleming and Adams, 1997; Byström, 1998). Economic literature has extensively covered water quantity and its quality, but separately, as illustrated by the apparent gap between joint quantity and quality management in these two branches of literature. Recently some studies focused on water quality and irrigation management (e.g. Dinar and Xepapadeas, 1998; Albiac et al., 2001). Palma (1999) introduced quality into a resource management model, but the model contains unrealistic simplifications with respect to the hydrological component.

The aim of this chapter is to study socially optimal agricultural groundwater extraction paths and to show how desiccation and contamination can be integrated into an optimal control model. In contrast to other approaches, our approach considers changes in both the quantity and the quality of the stock simultaneously, because these factors are interacting. Since we focus on the analytical aspects, the analysis remains theoretical. For an empirical application, insight is needed into the impact of extraction on crop growth and nature development. This

[6] Chapter 5 is a modified version of Hellegers, P.J.G.J, D. Zilberman and E.C. van Ierland, 2001, Dynamics of agricultural groundwater extraction. *Ecological Economics*, 37:303-311.

is, however, an indirect site-specific relationship, since groundwater extraction affects the availability of soil moisture in the root zones through changes in capillary rise. We use an interdisciplinary model, which shows the interaction between economic, hydrological and environmental variables. The model used in this chapter builds upon those developed by Caswell and Zilberman (1985 and 1986), Dinar and Zilberman (1991), Shah and Zilberman (1994) and Zilberman et al. (1994).

In this chapter the economic module of Figure 1.5 will be studied, i.e. how changes in groundwater extraction affect agriculture and nature as well as stock quantity and quality. Section 5.2 describes the setting of the agricultural groundwater extraction problem and shows how changes in stock quantity and quality can be modelled. It also examines the role of substitutable irrigation technologies. Section 5.3 shows the open access outcome and the socially optimal outcome and features of the optimal control model. Section 5.4 shows the importance of joint quantity and quality management. Section 5.5 contains the conclusions.

5.2 Dynamic joint quantity and quality management approach

To explain the impact of agricultural shallow groundwater extraction on groundwater quality and quantity, we started in Figure 1.2 with a schematic representation of water flows in the unsaturated and saturated zone of agricultural soils in the Netherlands. Figure 5.1 shows the groundwater stock $\boldsymbol{S}$ (m^3)[7] as a function of width x, length y and height z: $\boldsymbol{S} = xyz$. A smaller stock caused by agricultural shallow groundwater extractions, $\boldsymbol{E}$ (m^3/month), is associated with lower groundwater levels z (m) for a given area xy (m^2) which can cause the desiccation of nature. Only fraction h (-) of applied irrigation water is utilised by the crop, while the other fraction $(1-h)$ returns to the groundwater stock. Net natural groundwater recharge $\boldsymbol{R}$ (m^3/month) is equal to percolation $\boldsymbol{O}$ (m^3/month) minus capillary rise $\boldsymbol{G}$ (m^3/month) in that area during that time period: $\boldsymbol{R = (O\text{-}G)}$.

The setting of the problem is a phreatic aquifer on top of an aquitard, which can be considered as impenetrable. We assume that there is neither drainage nor inlet of foreign water and that there are no seepage flows. This means that the change in the groundwater stock in the saturated zone – which is the difference between in- and outgoing flows – is equal to the net natural recharge and return flows of irrigation water minus extraction. We also assume that nitrate is fully mixed in the aquifer, so that the groundwater stock is of a homogenous quality, which seems to be justified in the presence of a fully penetrating well.

[7] Bold symbols are expressed in m^3; to distinguish them from the same variable expressed in m, used in previous chapters.

In our analysis we do not consider de-nitrification processes in the soil. This simplification only holds on extremely mineral poor and organic poor sandy soils. In practice higher groundwater levels lead to more de-nitrification, which reduces nitrate leaching problems (Groeneveld et al., 1998).

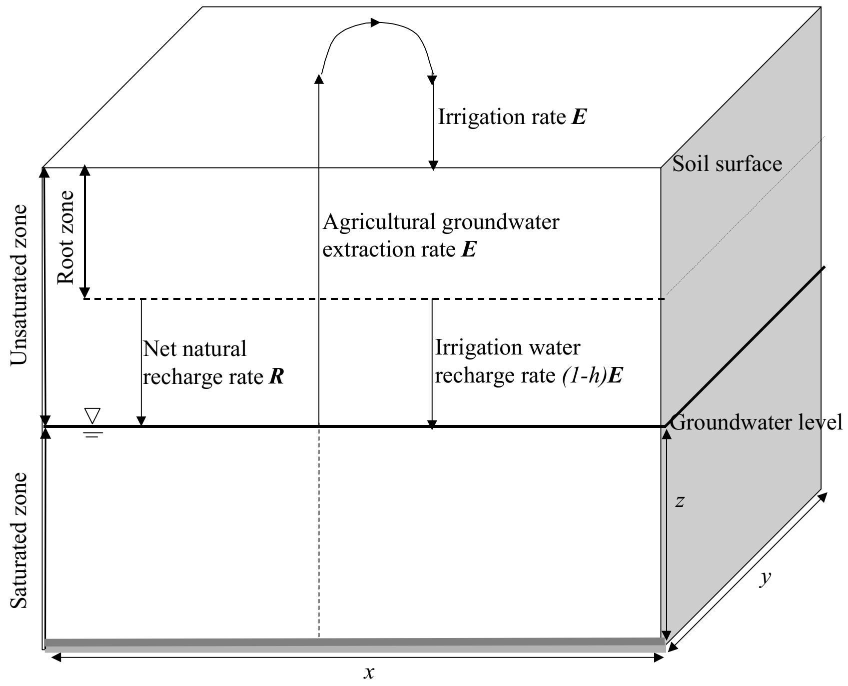

Figure 5.1 Schematic representation of the water flows in the unsaturated and the saturated zone.

The equations of motion of changes in groundwater stock quantity $\boldsymbol{S}$ and quality N^{S} for a given area are based on balance equations of what goes in and out of the stock[8]:

$$\frac{\partial \boldsymbol{S}}{\partial t} = \boldsymbol{R}_t + (1-h)\boldsymbol{E}_t - \boldsymbol{E}_t = \boldsymbol{R}_t - h\boldsymbol{E}_t \text{, for } \boldsymbol{S}_t \geq 0 \text{ and given initial condition } \boldsymbol{S}_0 \tag{5.1}$$

$$\frac{\partial N^{\mathrm{S}}}{\partial t} = (N_t^{\mathrm{R}} - N_t^{\mathrm{S}})\frac{(\boldsymbol{R}_t - h\boldsymbol{E}_t + \boldsymbol{E}_t)}{(\boldsymbol{R}_t - h\boldsymbol{E}_t + \boldsymbol{S}_t)} \text{, for } N_t^{\mathrm{S}} \geq 0 \text{ and given initial condition } N_0^{\mathrm{S}} \tag{5.2}$$

We continue with $\frac{\partial \boldsymbol{S}}{\partial t} \equiv \dot{\boldsymbol{S}}_t$ and $\frac{\partial N^{\mathrm{S}}}{\partial t} \equiv \dot{N}_t^{\mathrm{S}}$.

---

[8] For convenience, the derivation of the equation of motion of groundwater quality is in discrete time:

$N_{t+1}^{\mathrm{S}}\boldsymbol{S}_{t+1} = N_t^{\mathrm{S}}(\boldsymbol{S}_t - \boldsymbol{E}_t) + N_t^{\mathrm{R}}(\boldsymbol{R}_t - h\boldsymbol{E}_t + \boldsymbol{E}_t) \Rightarrow N_{t+1}^{\mathrm{S}}\boldsymbol{S}_{t+1} = N_t^{\mathrm{S}}\boldsymbol{S}_{t+1} + N_t^{\mathrm{S}}(\boldsymbol{S}_t - \boldsymbol{E}_t - \boldsymbol{S}_{t+1}) + N_t^{\mathrm{R}}(\boldsymbol{R}_t - h\boldsymbol{E}_t + \boldsymbol{E}_t) \Rightarrow$

$\dot{N}_t^{\mathrm{S}} = (N_t^{\mathrm{S}}(\boldsymbol{S}_t - \boldsymbol{E}_t - \boldsymbol{S}_{t+1}) + N_t^{\mathrm{R}}(\boldsymbol{R}_t - h\boldsymbol{E}_t + \boldsymbol{E}_t)) / \boldsymbol{S}_{t+1}$. Substitution by $\boldsymbol{S}_{t+1} = \boldsymbol{R}_t - h\boldsymbol{E}_t + \boldsymbol{S}_t$ gives equation (5.2).

Equation (5.1) indicates that changes in groundwater quantity over time $\dot{\boldsymbol{S}}_t$ for a given area are equal to net natural recharge flows $\boldsymbol{R}_\mathrm{t}$ plus recharge flows from applied water $(1-h)\boldsymbol{E}_\mathrm{t}$ that is not utilised by the crop minus agricultural extraction flows $\boldsymbol{E}_\mathrm{t}$ (all terms in (m^3/month)).

Equation (5.2) shows that changes in average groundwater quality over time $\dot{N}_t^\mathrm{S}$ for a given area consists of a quality and quantity component. As a quality indicator we use the nitrate concentration (g/m^3=mg/l). The average *quality* component is related to the difference between the nitrate concentration N_t^R in *recharge flows* and the nitrate concentration N_t^S in *extraction flows*. The latter is equal to the nitrate concentration in the groundwater stock. Groundwater quality will deteriorate if recharge flows are of a lower quality (higher nitrate concentration) than extraction flows: $N_t^\mathrm{R} > N_t^\mathrm{S}$. The nitrate concentration in recharge flows depends in our model on the concentration of nitrogen in the soil. For simplicity we have assumed that the nitrate concentration in recharge flows does not depend on the size of recharge flows. The quantity component shows that the smaller the ratio between recharge flows and groundwater stock, the larger the dilution effect and the smaller the change in stock quality. In the special case where the groundwater stock is very large, extraction has hardly any impact on stock quality.

The impact of irrigation technologies on the irrigation effectiveness

Farmers may apply various irrigation technologies that determine which fraction h of applied water is utilised by the crop; this is often referred to as the irrigation effectiveness of the technology. Changes in stock quantity and quality depend on the effectiveness of irrigation technologies. To show that increases in water and output prices provide incentives to adopt modern irrigation technology, we turn to the model of Zilberman and Lipper (1999). Output per hectare Y is, ceteris paribus, given by $Y=f(\boldsymbol{e})$, where $\boldsymbol{e}$ is effective water[9], which is defined as the amount of irrigation water actually used by the crop, with $f' > 0$ and $f'' < 0$, i.e. $f(\boldsymbol{e})$ is an increasing and concave agronomic function. The irrigation effectiveness h_j of technology j is the ratio between effective and applied water, according to $h_j=\boldsymbol{e}_j/\boldsymbol{E}_j$, and depends on land quality[10], which we hold constant for the sake of simplicity. Two irrigation

[9] In case effective water use is a function of water quality, groundwater of a lower quality will reduce effective water use. Effective water use is not a function of water quality in our analysis, because such use currently does not depend on the nitrate concentration in groundwater in the Netherlands.

[10] Land quality is defined in terms of the land's ability to store water and depends on soil permeability, water-holding capacity, and the slope of the land. Irrigation effectiveness is higher on heavier clay soils than on sandy soils, through which water passes rapidly. Differences in effectiveness are larger on sandy soils than on clay soils, and gains from a switch in technology will therefore be higher on sandy soils (cf. Shah et al., 1995).

technologies are considered: a traditional one $j=1$ and a modern one $j=2$, with a higher irrigation effectiveness $h_2>h_1$. The modern technology produces the same maximum output per hectare as the traditional technology $f(h_1\boldsymbol{E}_1)= f(h_2\boldsymbol{E}_2)$; however, although it requires less water $\boldsymbol{E}_2<\boldsymbol{E}_1$ since $h_2>h_1$, it involves higher investments $K_2>K_1$. Figure 5.2 shows the value of the marginal productivity *(VMP)* of both technologies. If applied water is smaller than $\boldsymbol{E}^*$, the value of the marginal product of the modern technology will be higher. Between $\boldsymbol{E}^*$ and $\boldsymbol{E}_1$, the value of the marginal product of the traditional technology will be higher. If the price of irrigation water is smaller than v^*, profit-maximising farmers will use the traditional technology, which requires more water to reach its maximum yield ($\boldsymbol{E}_1>\boldsymbol{E}_2$). If the price or irrigation water is larger than v^*, they will use the modern technology.

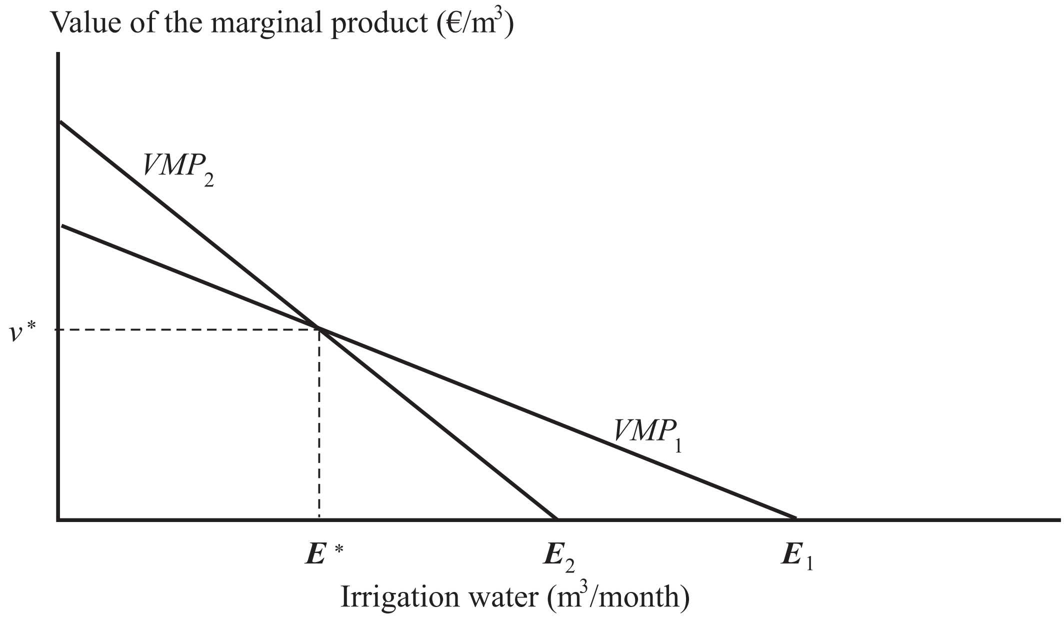

Figure 5.2 Marginal production value under alternative irrigation technologies.
Source: Zilberman et al. (1994).

Quasi-rent π per hectare is equal to agricultural output price P times output per hectare, minus the price of applied water v times the quantity of water applied $\boldsymbol{E}$ and the cost of technology K_j per hectare. Maximum competitive quasi-rent π_j^* is obtained by solving for the optimal level of applied water $\boldsymbol{E}_j^*$. The modern technology is chosen if $\pi_2^* > \pi_1^*$ and $\pi_2^* > 0$,

$$\pi_j^*(\boldsymbol{E}_j) = \max\{Pf(h_j\boldsymbol{E}_j) - v\boldsymbol{E}_j - K_j\}, \text{ for } j = 1,2 \tag{5.3}$$

The quasi-rent difference between the two technologies can be written as: $\Delta\pi=P\Delta q\text{-}v\Delta\boldsymbol{E}\text{-}\Delta K$. The modern technology will be selected in cases where the increased quasi-rents from higher yields or lower water cost offset higher investment cost. These results indicate that the adoption of modern irrigation technology will increase with increasing water or output prices (cf. Zilberman and Lipper, 1999). In other words, a higher price of water provides incentives to adopt modern irrigation technology. Under socially optimal applied irrigation water prices v^* and application levels $\boldsymbol{E}^*$, it is therefore more likely that modern technology will be adopted. Quasi-rent maximisation under technology j occurs where the value of the ma rginal product of effective water is equal to the price of effective water use.

$$Pf'(h_j\boldsymbol{E}_j) = \frac{v}{h_j}, \text{ for } j = 1,2 \tag{5.4}$$

The analysis now allows calculation of the open access outcome of groundwater extraction and the socially optimal results, where both quality and quantity aspects are considered.

5.3 Open access outcome versus socially optimal outcome

Open access outcome

If a large number of competitive farmers exploit a stock as a common property resource, it is not unreasonable to assume that their behaviour will be myopic. Individual farmers do not consider the impact of their extractions on the state of the resource and on the environment, and take the resource stock as given each period. Only their extraction costs k are considered in the price of applied water. Farmers will maximise individual current profit each period, and it seems reasonable to assume that they will pump water until the marginal net benefit is zero. Optimal groundwater use for a given technology at time t in the open access case is given by[11]:

$$Pf'(h\boldsymbol{E}_t) = \frac{k(\boldsymbol{S}_t)}{h} \tag{5.5}$$

The farmers will base their decisions only on the private cost and the resulting low price of water will provide fewer incentives to adopt modern irrigation technology than a price which reflects the social costs.

[11] This result is independent of discount rates. Under open access, equilibrium rents are zero, whatever discount rates are used, and a static analysis will therefore give correct results (Perman et al., 1999).

Socially optimal outcome

The objective of a social planner is to maximise the sum of discounted net agricultural benefit and environmental damage over an infinite time horizon, taking into account the changes in the quantity and quality of the groundwater stock over time. Shadow prices of changes in stock quantity and quality are considered in our continuous-time optimal control model. The level of damage to environmental amenities, given by $l = g(\boldsymbol{E}_t, N_t^{\mathrm{S}})$, is assumed to increase if farmers extract more shallow groundwater since groundwater levels will fall and if the nitrate concentration of the stock increases, with $g'(\boldsymbol{E}_t) > 0$ and $g'(N_t^{\mathrm{S}}) > 0$. The increase in damage becomes smaller for higher levels of extraction and higher nitrate concentrations, with $g''(\boldsymbol{E}_t) < 0$ and $g''(N_t^{\mathrm{S}}) < 0$. The unit cost of groundwater extraction $k(\boldsymbol{S}_t)$ increases as the size of the stock S_t declines, and the cost increase per unit is larger, the lower the remaining stock, with $k' < 0$ and $k'' > 0$, i.e. $k(\boldsymbol{S}_t)$ is decreasing and convex. A small stock increases the unit cost of extraction and provides an incentive to reduce groundwater extraction. Further we assume a constant discount rate ρ. Finally, we define V as the annual monetary value of nature per hectare and Φ as the ratio between the area of affected nature reserve and the area of farmland irrigated. To maximise the total present value of the objective function, the social planner's problem is to choose $\boldsymbol{E}_t$ for a given technology:

$$\max \int_0^\infty (pf(h\boldsymbol{E}_t) - \Phi V g(\boldsymbol{E}_t, N_t^{\mathrm{S}}) - k(\boldsymbol{S}_t)\boldsymbol{E}_t) e^{-\rho t} \mathrm{d}t \qquad (5.6)$$

subject to the equations of motion (equations (5.1) and (5.2)) of the two state variables. The maximum principle technique is used to solve the optimisation problem (Perman et al., 1999).

The current value Hamiltonian function for the optimisation problem can be stated as:

$$H = pf(h\boldsymbol{E}_t) - \Phi V g(\boldsymbol{E}_t, N_t^{\mathrm{S}}) - k(\boldsymbol{S}_t)\boldsymbol{E}_t + \nu_t(\boldsymbol{R}_t - h\boldsymbol{E}_t) - \mu_t(N_t^{\mathrm{R}} - N_t^{\mathrm{S}})\frac{(\boldsymbol{R}_t - h\boldsymbol{E}_t + \boldsymbol{E}_t)}{(\boldsymbol{R}_t - h\boldsymbol{E}_t + \boldsymbol{S}_t)} \qquad (5.7)$$

where ν_t and μ_t are the current value shadow prices or co-state variables associated with changes in the quantity and quality of the resource over time, i.e. the values of respectively a unit change in both the availability and the nitrate concentration of the groundwater stock at time t (cf. Conrad and Clark, 1987). Optimal allocation rules are given by:

$$\frac{\partial H}{\partial \boldsymbol{E}_t} = Pf'(h\boldsymbol{E}_t)h - \Phi Vg'(\boldsymbol{E}_t) - k(\boldsymbol{S}_t) - v_t h - \frac{\mu_t(N_t^{\mathrm{R}} - N_t^{\mathrm{S}})(\boldsymbol{R}_t - h\boldsymbol{S}_t + \boldsymbol{S}_t)}{(\boldsymbol{R}_t - h\boldsymbol{E}_t + \boldsymbol{S}_t)^2} = 0 \tag{5.8}$$

$$\dot{v}_t = \rho v_t - \frac{\partial H}{\partial \boldsymbol{S}_t} = \rho v_t + k'(\boldsymbol{S}_t)\boldsymbol{E}_t - \frac{\mu_t(N_t^{\mathrm{R}} - N_t^{\mathrm{S}})(\boldsymbol{R}_t - h\boldsymbol{E}_t + \boldsymbol{E}_t)}{(\boldsymbol{R}_t - h\boldsymbol{E}_t + \boldsymbol{S}_t)^2} \tag{5.9}$$

$$\dot{\mu}_t = \rho\mu_t - \frac{\partial H}{\partial N_t^{\mathrm{S}}} = \rho\mu_t + \Phi Vg'(N_t^{\mathrm{S}}) - \frac{\mu_t(\boldsymbol{R}_t - h\boldsymbol{E}_t + \boldsymbol{E}_t)}{(\boldsymbol{R}_t - h\boldsymbol{E}_t + \boldsymbol{S}_t)} \tag{5.10}$$

The first optimality condition (equation (5.8)) can be rewritten as:

$$Pf'(h\boldsymbol{E}_t) = \frac{\Phi Vg'(\boldsymbol{E}_t)}{h} + \frac{k(\boldsymbol{S}_t)}{h} + v_t + \frac{\mu_t(N_t^{\mathrm{R}} - N_t^{\mathrm{S}})(\boldsymbol{R}_t - h\boldsymbol{S}_t + \boldsymbol{S}_t)}{h(\boldsymbol{R}_t - h\boldsymbol{E}_t + \boldsymbol{S}_t)^2} \tag{5.11}$$

To achieve socially optimal agricultural groundwater extraction, the value of the marginal damage to environmental amenities, the extraction costs and shadow prices of changes in the quantity and quality of the stock over time have to be considered in the price of water use. Agricultural groundwater extraction will have a negative impact on groundwater quality if $N_t^{\mathrm{R}} > N_t^{\mathrm{S}}$ and a positive impact if $N_t^{\mathrm{R}} < N_t^{\mathrm{S}}$. The price of water will be higher in the first case, when extraction causes a negative externality. The significance of this impact on the price of water will become smaller if stock size increases, due to the dilution effect.

The rate of change over time in shadow prices can be obtained from equations (5.9) and (5.10).

$$\frac{\dot{v}_t}{v_t} = \rho + \frac{k'(\boldsymbol{S}_t)\boldsymbol{E}_t}{v_t} - \frac{\mu_t(N_t^{\mathrm{R}} - N_t^{\mathrm{S}})(\boldsymbol{R}_t - h\boldsymbol{E}_t + \boldsymbol{E}_t)}{v_t(\boldsymbol{R}_t - h\boldsymbol{E}_t + \boldsymbol{S}_t)^2} \tag{5.12}$$

$$\frac{\dot{\mu}_t}{\mu_t} = \rho + \frac{\Phi Vg'(N_t^{\mathrm{S}})}{\mu_t} - \frac{(\boldsymbol{R}_t - h\boldsymbol{E}_t + \boldsymbol{E}_t)}{(R_t - h\boldsymbol{E}_t + \boldsymbol{S}_t)} \tag{5.13}$$

The rate of change in the resource value associated with delayed extraction by one period (equation (5.12)) (i.e. the cost of not mining the resource) is equal to the sum of three effects:

1) the discount rate, which is positive and serves as a compensation for delayed benefits;
2) the extraction-cost effect, where larger stocks reduce the extraction cost; and
3) the dilution effect, where larger stocks tend to slow down changes in quality. The dilution effect will reduce the cost of maintaining stocks if $N_t^{\mathrm{R}} > N_t^{\mathrm{S}}$ and it will increase the cost if $N_t^{\mathrm{R}} < N_t^{\mathrm{S}}$. In the latter case, water quality is not improved due to delayed extraction.

If the initial stock size is relatively large, the extraction-cost and dilution effect may be negligible, because a marginal change in stock quantity is unlikely to cause a substantial change, either in the unit extraction cost or in the stock quality. In that case, the rate of change in the shadow price of stock quantity will be equal to the discount rate. If the initial stock size is small relative to recharge flows (i.e. if the extraction-cost and the dilution effect are stronger than the discount-rate effect), the rate of change in the shadow price will decline over time because the extraction-cost and dilution effect will become stronger over time.
The rate of change in the shadow price of stock quality over time (equation (5.13)) is also equal to the sum of three effects: 1) the discount rate, 2) the environmental damage-effect, which is positive (higher nitrate concentrations increase costs) and 3) the dilution effect.

We should like to emphasise that the rate of change over time in shadow prices (equations (5.12) and (5.13)) differ from the results of the dynamics of renewable resource economics found in the literature (Zilberman et al., 1993). In the literature, the rate of change in the resource value associated with delayed extraction by one period generally depends on a resource-growth effect (where maintaining stocks tends to increase resource growth), rather than on a dilution effect as in our analysis. When the resource is a population of a livestock species, for instance a fish population, population growth depends on the initial population size (reflected in the growth function). Insight into the importance of stock size in slowing down changes in stock quality is therefore an extension of existing work in this field.

Usually, inter-temporal optimisation models are closed by adding some sort of terminal condition. If there is no recharge, the stock will always be decreasing, so the problem will necessarily reach a point where nothing will be extracted, because either the stock is depleted or extraction has become prohibitively expensive. If groundwater is a renewable resource, it is possible to have a steady state with extraction, for which: $\dot{\boldsymbol{S}} = 0$, $\dot{N}^{\mathrm{S}} = 0$, $\dot{\lambda} = 0$, and $\dot{\mu} = 0$. If the quantity and quality of the stock do not change over time, and consequently shadow prices remain constant, a renewable resource system will be in a steady state. The results are expressed in the following equations:

$$\boldsymbol{E} = \frac{\boldsymbol{R}}{h} \tag{5.14}$$

$$\frac{(N^{\mathrm{R}} - N^{\mathrm{S}})(\boldsymbol{R} - h\boldsymbol{E} + \boldsymbol{E})}{(\boldsymbol{R} - h\boldsymbol{E} + \boldsymbol{S})} = 0 \tag{5.15}$$

$$\nu = -\frac{k'(\boldsymbol{S})\boldsymbol{E}}{\rho} + \frac{\mu(N^{\mathrm{R}} - N^{\mathrm{S}})(\boldsymbol{R} - h\boldsymbol{E} + \boldsymbol{E})}{\rho(\boldsymbol{R} - h\boldsymbol{E} + \boldsymbol{S})^2} \tag{5.16}$$

$$\mu = -\frac{\Phi V g'(N^{\mathrm{S}})}{\rho - (\boldsymbol{R} - h\boldsymbol{E} + \boldsymbol{E})/(\boldsymbol{R} - h\boldsymbol{E} + \boldsymbol{S})} \tag{5.17}$$

If extraction and recharge flows are of the same size, the stock size will not change over time (equation (5.14)). Neither will stock quality change over time, if nitrate concentrations in recharge and extraction flows are equal $N^{R} = N^{S}$, if there is no recharge $\boldsymbol{R} - h\boldsymbol{E} + \boldsymbol{E} = 0$, or if stock size is very large $\boldsymbol{S} \to \infty$, (equation (5.15)). The shadow price of changes in the quantity of the resource over time will be smaller for larger stocks (equation (5.16)). The shadow price of changes in quantity are zero, if stock size is very large and a quality-only model will be appropriate. The smaller the ratio between the recharge flows and the groundwater stock, the smaller the shadow price of changes in quality of the resource (equation (5.17)).

5.4 Importance of incorporating water quality into a resource management model

In this section, we show the importance of introducing the impact of groundwater extraction on groundwater quality into a resource management model. We study water-pricing reform, a key element of the European Water Framework Directive (COM(2000)60)[12] in the presence of negative and positive externalities from agricultural groundwater extraction on stock quality. Such positive externalities may arise if N_t^{R} becomes smaller than N_t^{S}, which might for instance be the result of current restrictions on the maximum allowable concentrations of nitrates. According to the Nitrate Directive (Council Directive 91/676/EEC), waters must be protected against pollution by nitrates from agricultural sources by not allowing the nitrate concentration in groundwater for drinking water purposes to exceed the legally accepted EU limit of 50 mg/l.

Currently in the Netherlands, most farmers only pay the energy costs of lifting water from the stock to the field (about € 0.04 per m^3) although extraction is regulated by two acts. Under the 1995 *Taxes on an Environmental Basis Act* agricultural groundwater extraction is subject to a tax (of € 0.08 per m^3), but only a small percentage of farmers (about 2%) exceeds the tax-free threshold of 40,000 m^3 of groundwater extraction per annum (van Staalduinen et al., 1996). Under the 1983 *Groundwater Act,* agricultural groundwater extraction is subject to a levy. The tariffs and levy-free threshold vary among provinces. The levy is relatively low compared to the tax. The main part of agricultural groundwater extraction is, however, not subject to the levy under the *Groundwater Act*. This means that the price of irrigation water

[12] Recovery of costs for water, including environmental and resource costs is part of the European Water Framework Directive (COM(2000)60), which is the legislative document that will guide European water policies. The directive requires Member States to undertake economic analysis of water use (Article 5 and Annex III) and to ensure that by 2010 water pricing policies provide adequate incentives to use water efficiently (Article 9).

is currently equal to the price in the open access case. Such a low price is inefficient from a social point of view in the presence of externalities such as desiccation and contamination and provides fewer incentives to adopt modern irrigation technology than optimal. To achieve socially optimal agricultural groundwater extraction paths, the costs of these externalities should to be internalised in the price of water.

Article 12 of the European Water Framework Directive will oblige member states to implement 'full cost recovery', which means that the price of water should reflect not only the costs of the water-use services, but also the environmental and resource depletion costs. This will provide incentives to adopt modern irrigation technology. Whether modern technology will be adopted depends on, among other things, the gap between relative costs of both irrigation technologies, as explained in Section 5.2. The extent of divergence between the private and the social price of water (equations (5.5) and (5.11)) represents the optimal volumetric tax T^{v} that will induce farmers to behave in the socially optimal way:

$$T^{\mathrm{v}} = \frac{\Phi V g'(\boldsymbol{E}_t)}{h} + v_t + \frac{\mu_t (N_t^{\mathrm{R}} - N_t^{\mathrm{S}})(\boldsymbol{R}_t - h\boldsymbol{S}_t + \boldsymbol{S}_t)}{h(\boldsymbol{R}_t - h\boldsymbol{E}_t + \boldsymbol{S}_t)^2} \tag{5.18}$$

Agricultural groundwater extraction will have a negative impact on groundwater quality if $N_t^{\mathrm{R}} > N_t^{\mathrm{S}}$ and a positive impact if $N_t^{\mathrm{R}} < N_t^{\mathrm{S}}$. Contamination is reflected in a higher required tax on agricultural groundwater extraction and improvements in water quality are reflected in a lower required tax. The significance of this impact on the tax rate will become smaller if stock size increases, due to the dilution effect. If the negative externalities of extraction are not internalised in the tax, contamination will be accelerated because the price will be too low and extraction may be higher than optimal. If the positive externalities of extraction are not internalised in the tax, improvements in quality will be slowed down because the price will be too high, which may decrease extraction and increase stock size and the irrigation effectiveness. In other words, it will affect the time path of changes in stock quality.

Larson et al. (1996) suggest that water may be the better input to regulate in terms of achieving non-point source reduction goals at lower cost, whereas often only a tax on nitrogen input is considered in the analysis for efficient pollution regulation, as in Fleming and Adams (1997). The cost-effectiveness of second-best policies in achieving joint quantity and quality management can be evaluated along the lines of Larson et al. (1996).

The theoretical framework of efficient water-pricing schemes is clear, but there are some caveats. Firstly, it is hard to determine the level of taxes, since monetary values have to be attached to damage caused by excessive use of groundwater, whereas perpetrators of externalities usually evaluate damage less severely than other interest groups. Solutions suggested for the monetary valuation of environmental damage caused by excessive groundwater extraction are very controversial, which makes the direct application of equations (5.11) and (5.18) debatable. Secondly, water-pricing schemes often ignore the information needed for implementation. Implementation problems are linked to enforcement, monitoring, institutional limitations, conflicting policies, political interests and welfare implications. Thirdly, the introduction of price reform is conditional upon the size of social gains relative to transaction costs. Finally, water-pricing reform has a positive influence on water conservation only if the price elasticity of water demand is significantly different from zero and is negative.

In this chapter we only focused on water-pricing reform as an instrument in achieving socially optimal agricultural groundwater extraction paths. However, other kinds of policy instruments can be used as well. In Section 6.4 of this thesis we will discuss the suitability of various policy instrument for groundwater extraction management.

5.5 Conclusions

Although some simplifying assumptions are made, the analysis does show the importance of bringing the impact of agricultural shallow groundwater extraction on groundwater quality into a resource management model. It studies the dynamics of socially optimal agricultural shallow groundwater extraction management. This is not only of interest in the Netherlands, but also in other countries with a comparable hydrological setting and similar problems.

It appears that the current low price of agricultural groundwater use is inefficient and provides fewer incentives to adopt modern irrigation technology than a system that considers the cost of desiccation and contamination in the price of water does.

Internalisation of the negative as well as the positive externalities from agricultural shallow groundwater extraction on stock quality in the price of groundwater is particularly significant if the recharge of groundwater is large compared to stock size. This impact will become smaller if stock size increases, due to the dilution effect. If these externalities are not internalised in the price of groundwater, contamination will be accelerated and quality improvements will be slowed down. It will affect the time path of changes in groundwater stock quality over time. This stresses the importance of considering developments in nitrate policy when designing water-price reforms, which is a key element of the European Water Framework Directive.

6 Policy instruments for groundwater management for agriculture and nature[13]

6.1 Introduction

In the presence of growing conflicts between agriculture and nature with respect to groundwater management, there is a pressing need to understand the potential role of policy instruments. There is, however, a lack of insight into the suitability of policy instruments to achieve optimal groundwater level and extraction management in the Netherlands.

Economic literature on the role of policy instruments for groundwater management in the Netherlands is limited, especially with respect to agriculture. Wiersma (1998) investigated a more efficient groundwater allocation procedure based on regional auctions for Dutch groundwater extraction permits. However, he does not consider agricultural extractions, because they are only temporary and will be reduced by sprinkling bans. Extensive literature on the role of groundwater allocation mechanisms in other countries is available (e.g. Michelsen and Young 1993; Dinar and Wolf 1994; Strosser 1997), but this literature often refers to semi-arid areas and the typical case of water scarcity.

As policy instruments have major impacts on the efficiency and effectiveness of water allocation as well as on the distribution of costs and benefits, which may affect the political feasibility, it is important to study both their current and their potential role. The aim of this chapter is therefore to test the suitability of policy instruments to reduce the desiccation of nature, which depends on a large number of specific characteristics, such as geohydrological characteristics and the cause of the desiccation problem (kind of hydrological change).

The desiccation of nature caused by the intensified drainage of agricultural areas is the result of conflicting interests at stake with respect to the management of the *in situ* services of groundwater. The desiccation of nature caused by increased extraction is the result of extraction on the *in situ* services of groundwater. Groundwater levels can be managed by controlling drainage. Groundwater extraction can be managed by influencing demand.

[13] Chapter 6 is an extended version of Hellegers, P.J.G.J. and E.C. van Ierland, 2001, Policy instruments for groundwater management for agriculture and nature in the Netherlands. In: P. Pashardes, T. Swanson and A. Xepapadeas (eds.), *Economics of Water Resources: Theory and Policy.* Kluwer Publishers, Dordrecht.
Section 6.3 is a modified version of Hellegers, P.J.G.J. and E.C. van Ierland, forthcoming, Policy instruments for groundwater allocation in Dutch agriculture. *Water Resources Update*, 00:000-000.

Groundwater policy instruments and their advantages and disadvantages are discussed separately in this chapter. Policy instruments can, however, be combined in such a way that they reinforce each other, which brings OECD (1991) to conclude that a mixture of policy instruments can be very fruitful. There are many cases where economic instruments are applied in conjunction with other instruments; especially combinations with regulatory instruments are quite common. Water can for instance be diverted away from agriculture by means of a groundwater extraction quota, while encouraging trading by means of tradable groundwater extraction rights. General guidelines for the use of policy instruments are, however, difficult to establish, since the suitability of policy instruments depends on the characteristics of the situation. An empirical analysis of the region-specific circumstances is necessary in order to provide insight into the optimal policy mix.

In this chapter the third module of Figure 1.5 is studied: the policy module. In Section 6.2 the efficiency gains and transaction costs of policy reform are studied in an analytical way, since policy reform is usually conditional upon the size of efficiency gains relative to transaction costs involved. In Section 6.3 the current instruments for groundwater level management are discussed and the suitability of alternative instruments is tested against six policy review criteria (discussed in Section 2.4). In Section 6.4 current instruments for groundwater extraction management are discussed and the suitability of alternative instruments is tested. Section 6.5 contains the conclusions.

6.2 Transaction costs

Policy reform is usually conditional upon the size of efficiency gains relative to transaction costs, i.e. implementation, enforcement and monitoring costs. Transaction costs diminish the net benefits of reforms. Zilberman et al. (1997) showed that the transition from a water queuing system to a tradable water rights system may lower social welfare in cases of high transaction costs. This occurs when potential gains are lost to search and negotiation costs. Zhang (2001) also stresses the importance of transaction costs. He notes that although benefits of change from open access to regulated access, as a remedy for ‘the tragedy of the commons’ has been widely recognised, the transaction costs are often neglected. Unfortunately, little empirical work has been done to determine the magnitude and form of transaction costs or what factors influence transaction costs. On the one hand, some fixed costs are likely to be associated with transactions (e.g. legal costs and registration fees), which suggests declining marginal costs. On the other hand, search costs are more likely to be characterised by rising marginal costs as potential traders first search for those traders who are most willing to trade (Renzetti, 2000). Because transaction costs are so important, the choice of institutional arrangement for dealing with these costs is critical.

In this section we will study (1) how the distribution of transaction costs between agriculture and nature affects the optimal agricultural groundwater extraction reduction level; and (2) how the distribution of transaction costs among farms affects the efficiency gains of a transition from a groundwater extraction quota to an extraction tax.

1)The impact of transaction costs on the agricultural groundwater extraction reduction level
Suppose that the marginal cost of extraction reduction to agriculture increase as more extraction is reduced, while the marginal costs to nature decline as extraction is reduced further. Figure 6.1 shows that if agricultural groundwater extraction is reduced from point A to Q^*, the costs to nature decrease (area ABCQ*), while the costs to agriculture increase (area ADCQ*). The difference (area BCD) reflects the efficiency gains of reducing extraction from point A to Q^*. Point Q^* is the optimal reduction level in terms of economic efficiency.

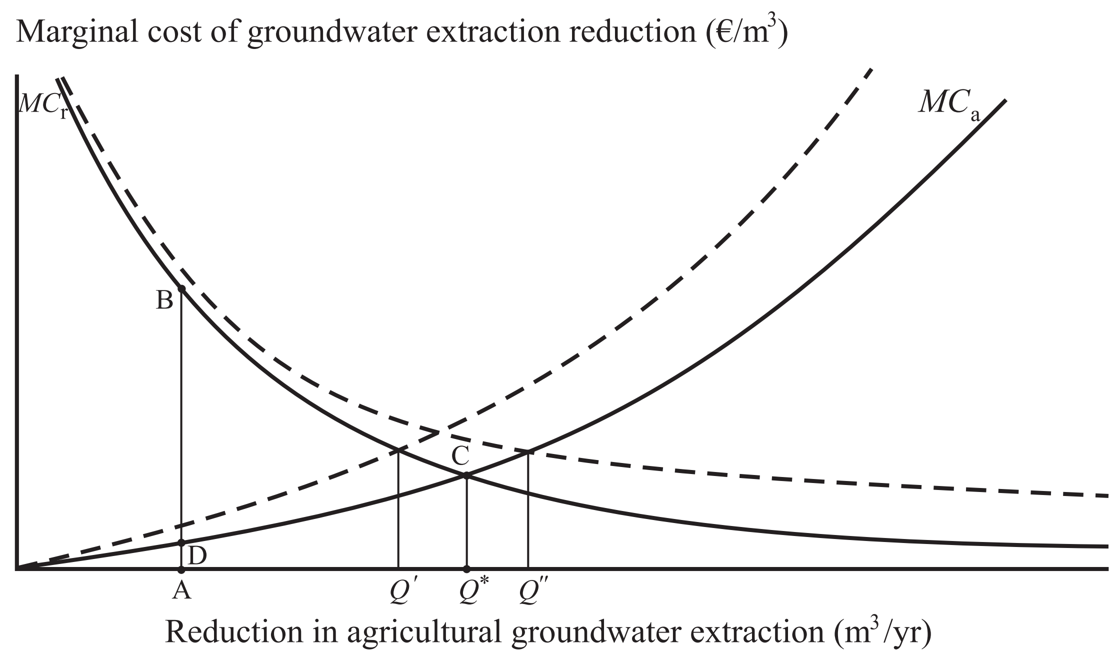

Figure 6.1 Marginal costs curves of extraction reduction to agriculture MC_a and nature MC_r without transaction costs (solid lines) and including marginal transaction costs (dashed lines).

To study the impact of transaction costs on agricultural groundwater extraction reduction, we derive the efficient groundwater extraction level without and with transaction cost. Suppose the marginal cost curves of extraction reduction to agriculture and nature can be written as $MC_a = \alpha Q^2$ and $MC_r = \beta Q^{-2}$ respectively.

a) Without transaction costs, the efficient groundwater extraction level (Figure 6.1 point Q^*) can be derived by equalising the marginal costs of extraction reduction to agriculture and nature:

$$\alpha Q^2 = \beta Q^{-2} \text{ which gives} \tag{6.1}$$

$$Q = \sqrt[4]{\frac{\beta}{\alpha}} \tag{6.2}$$

b) In the presence of transaction costs, which are rising with an increase in extraction reduction, the optimal groundwater extraction reduction level depends on the distribution of transaction costs. Assume that marginal transaction costs MT^c can be written as $MT^c = \delta Q^2$.

-If transaction costs are fully passed on to agriculture, costs are minimised when:

$$\alpha Q^2 + \delta Q^2 = \beta Q^{-2} \text{ which gives} \tag{6.3}$$

$$Q = \sqrt[4]{\frac{\beta}{(\alpha + \delta)}} \tag{6.4}$$

-If transaction costs are fully passed on to nature, costs are minimised when:

$$\alpha Q^2 = \beta Q^{-2} + \delta Q^2 \text{ which gives} \tag{6.5}$$

$$Q = \sqrt[4]{\frac{\beta}{(\alpha - \delta)}} \tag{6.6}$$

This means that the distribution of transaction costs affects the efficient extraction reduction level. The reduction in agricultural groundwater extraction decreases (Figure 6.1 point Q') when transaction costs are passed on to agriculture (equation (6.4)), since marginal costs (including marginal transaction costs) to agriculture of extraction reduction increase. The reduction in agricultural groundwater extraction increases (Figure 6.1 point Q'') when transaction costs are passed on to nature (equation (6.6)), since marginal cost (including marginal transaction costs) to nature of extraction reduction increase.

2) The efficiency gains of a transition from an extraction quota towards an extraction tax

In case farms have different production processes and different extraction reduction costs, their marginal costs of a reduction in extraction are not the same. Figure 6.2 shows different marginal extraction reduction cost curves for two farms: $MC_1 = \alpha_1 Q_1^2$ and $MC_2 = \alpha_2 Q_2^2$. In case farms are near each other, the marginal social benefit of a reduction in groundwater extraction is, however, the same, no matter which farm reduces its extraction. Suppose we want to reduce total groundwater extraction by $2Q_b$ in periods of serious droughts and that both farms face the same quota of Q_b. This will not be efficient, because farm 2 can reduce extraction more cheaply than farm 1. For different marginal cost curves of groundwater extraction reduction ($\alpha_1 \neq \alpha_2$), it is not efficient to reduce extraction by the same quota ($Q_1 = Q_2$), since in that case marginal costs are not equal.

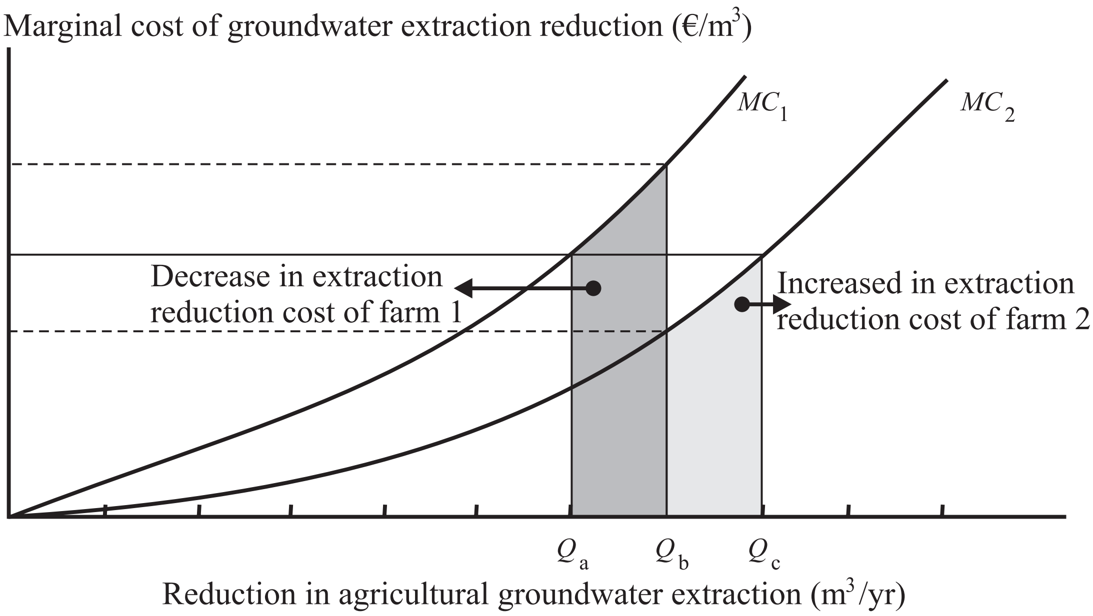

Figure 6.2 Efficiency gains of a transition from an extraction quota to an extraction tax.

Figure 6.2 shows that the most efficient way of reducing extraction by $2Q_b$ is to have farm 1 reduce extraction by Q_a and farm 2 reduce extraction by Q_c. Both farms have equal marginal costs of reduction of *v* in that case. The decrease in marginal costs of extraction reduction of farm 1 (dark grey area) exceeds the increase in marginal costs of extraction reduction of farm 2 (light grey area). The efficiency gains *g* of a transition from an extraction quota to an extraction tax is equal to the reduced costs to farm 1 minus the additional costs to farm 2:

$$g = \int_{Q=Q_a}^{Q=Q_b} \alpha_1 Q^2 \mathrm{d}Q - \int_{Q=Q_b}^{Q=Q_c} \alpha_2 Q^2 \mathrm{d}Q = \frac{\alpha_1}{3}(Q_b^3 - Q_a^3) - \frac{\alpha_2}{3}(Q_c^3 - Q_b^3) \qquad (6.7)$$

If total transaction costs of such policy reform, T^c, exceed these efficiency gains: $T^c > g$, quotas are preferred over taxes[14]. It depends on the distribution of transaction costs between both farms how both marginal cost curves (including transaction costs) will change. The larger the share of transaction costs a farm has to bear, the smaller its reduction in groundwater extraction, since it becomes more expensive to reduce extraction (the marginal cost curve becomes steeper when marginal transaction costs are included). The size of the efficiency gains of a tax instead of a quota depends on the slopes of both marginal cost curves. Efficiency gains will increase if the divergence between the slopes increases.

-*In the case of a quota* (without transaction costs), groundwater extraction will not be reduced in an efficient way. Groundwater extraction will be reduced to:

$$Q_1 = Q_2 = \sqrt[4]{\frac{\beta}{4(\alpha_1 + \alpha_2)}} \tag{6.8}$$

which can be derived by substitution of $Q_1 = Q_2$ into $\alpha_1 Q_1^2 + \alpha_2 Q_2^2 = \beta\left(Q_1 + Q_2\right)^{-2}$.

-*In the case of a tax* (without transaction costs), groundwater extraction will, however, be reduced in an efficient way. Both efficient groundwater extraction reduction levels can be derived by equalising marginal costs of extraction reduction to agriculture and nature (equation (6.9)) and marginal costs of extraction reduction of both farms (equation (6.10)):

$$\alpha_1 Q_1^2 + \alpha_2 Q_2^2 = \beta\left(Q_1 + Q_2\right)^{-2} \text{ and} \tag{6.9}$$

$$\alpha_1 Q_1^2 = \alpha_2 Q_2^2 \text{ which gives} \tag{6.10}$$

$$Q_1 = \sqrt[4]{\frac{\alpha_2 \beta}{2(\alpha_1 + \sqrt{\alpha_1 \alpha_2})^2}} \text{ and} \tag{6.11}$$

$$Q_2 = \sqrt[4]{\frac{\alpha_1^2 \beta}{2\alpha_2(\alpha_1 + \sqrt{\alpha_1 \alpha_2})^2}} \tag{6.12}$$

These optimal agricultural groundwater extraction reduction levels can be achieved by imposing a groundwater extraction tax, so that the price of agricultural groundwater extraction v becomes equal to the marginal cost of extraction $v = \alpha_1 Q_1^2 = \alpha_2 Q_2^2$.

[14] Quotas are also preferred to taxes if the government has limited information about the costs and benefits of extraction reduction, and the marginal cost curve of nature is steep and the marginal cost curve of agriculture is relatively flat. An error in setting a tax would result in that case in high social costs compared to the same percentage error in setting the quota (Pindyck and Rubinfeld, 1989).

6.3 Suitability of policy instruments for groundwater level management

The performance of six policy instruments for groundwater level management will be discussed in this section. First we describe the prerequisites for instruments to be effective. After that we discuss whether it improves the economic and technical efficiency. Next we discuss the size of transaction costs of policy instruments, which reflects the administrative feasibility. We also discuss whether the financial impact is equitably distributed among parties, which affects the acceptability. Finally, practical implementation problems will be addressed. The results of these tests are summarised in Table 6.1 with the aid of a (+,-) table.

Table 6.1 Suitability of instruments for groundwater level management [a].

Instrument	Effective-ness	Economic efficiency	Technical efficiency	Transaction costs	Equity	Acceptability
1A Institutional change	+	+	+	+/-	+	+
1B Tax	+/-	+	+	-	+/-	-
1C Tradable rights	+	+	+	-	+/-	+/-
1D Subsidy	+/-	-	-	+/-	-	+
1E Standard	+	-	-	+	+/-	-
1F Agreements	+/-	+	+	+/-	+	+/-

a) Three categories are distinguished: '+' stands for a positive impact, '-' for a negative impact and '+/-' indicates the possibility of a positive as well as a negative impact. A '+' can be interpreted as follows:
'+' for environmental effectiveness means that the instrument reaches its environmental objective;
'+' for economic efficiency indicates that the value of the marginal product of water is equal among users;
'+' for technical efficiency means that the instrument provides incentives to adopt modern technologies;
'+' for transaction costs means that the instrument is easy to implement and monitor against low cost;
'+' for equity means that the costs and benefits are equitably distributed among affected parties;
'+' for acceptability means that affected parties accept the instrument without serious resistance.

1A) *Institutional change* refers here to a change in the system that balance the interests with respect to water level management. Until 1992, agricultural interests dominated within the governing body of water boards due to the strong financial interest of this sector. Since 1992, environmental and nature organisations have been better represented in these governing bodies, which implies a transition to a *multi-interest management system*. Interests of nature with respect to groundwater level management have often been ignored in the decisions about surface water levels, which directly affect groundwater levels. If water boards become responsible for groundwater level management and if multi-interests are represented within the water boards, interests with respect to groundwater levels will be better balanced and can become guiding for surface water level management. This might improve the economic and technical efficiency in an equitable manner. Transaction costs seem to be low, although such changes are generally not so easy to implement. The acceptability seems no problem as long as the basic principle of water boards ('interest, payment and authority') is maintained.

1B) Theoretically, a *tax* can be used to discourage farmers from lowering the groundwater level below a certain threshold. The effectiveness of a tax depends on the right estimation of the marginal tax level and on the risk attitude of farmers with respect to wet damage. A differentiated tax level has to be created where there are local differences in the value of reserves or the vulnerability of the environment to changes in the groundwater level. An advantage of a tax is that it improves both economic and technical efficiency. The financial impact on affected parties depends on the restitution of revenues. Usually serious resistance is raised against the introduction of a tax. Transaction cost will be high, since a differentiated tax is not easy to control and monitor and due to practical implementation problems. Firstly, it is hard to define a good tax base. For instance, it is not easy to measure the amount of drainage water. A tax on a change in the groundwater level is also complicated, because external factors also affect the groundwater level. Secondly, it is also complicated to take account of heterogeneity in vulnerability of areas to changes in drainage. Finally, groups of farmers will be affected by changes in groundwater levels, which means that the decision of one farmer to raise the groundwater level will affect other farmers. These interdependencies make the internalisation of externalities complicated. Charging water boards for lowering surface water levels will not influence individual farmer’s behaviour, although it will affect the strategy of groups of farmers represented in the governing body of water boards.

1C) *Tradable rights* to lower the groundwater level are ceilings on lowering that, once initially allocated, can be traded subject to a set of prescribed rules. The environmental objective is the starting point. Tradable rights improve economic and technical efficiency, since the market determines the price of the right in a dynamic way (Pearce and Turner, 1990). The high transaction costs are a major disadvantage. The financial impact on affected parties as well as the acceptability depends on the initial allocation of rights. The use of tradable rights seems to be complicated in practice, since the impact of changes in the groundwater level on agriculture and nature depends on location-specific circumstances. To avoid the transfer of rights to areas sensitive to desiccation, trading among areas has to be restricted. Here we face a dilemma: on the one hand the market approach is embraced, but on the other hand we need a trade institution for guided trading (Kruitwagen et al., 2000).

1D) A *subsidy* is a reward for meeting a certain groundwater level which is higher than the desired standard. Subsidies are not economically efficient, they are disturbing and do not provide incentives to adopt modern technologies. The acceptability is no problem, since participation in subsidy schemes is voluntary and because of the financial implications. Implementation problems are similar to those of a tax. Farmers currently receive payments for drastic income losses due to higher groundwater levels. These payments are called subsidies, which is a misleading, since these payments do not provide incentives to lower the groundwater level, but are intended to balance the financial impacts on affected parties.

Whether this is justified depends on the allocation of rights to lower groundwater levels. Since farmers currently receive financial compensation for the private benefits they have foregone, the rights to lower groundwater levels have so far been ascribed to agriculture. These payments prevent farmers leaving the region due to income losses. In that respect it provides incentives to individual farmers.

1E) A legal groundwater level *standard* can be introduced. Such will be effective if farmers face substantial monetary penalties for lowering the groundwater level below this standard. The standard does not improve the economic efficiency and does not provide incentives to innovate. The financial impact is not always equitably distributed among the affected parties, since there are differences in the vulnerability of areas. The transaction costs of a standard are relatively low. Usually there is serious resistance to the introduction of standards.

1F) Currently *voluntary agreements* concerning drainage control are established between farmers and governmental organisations representing the interests of nature. Participation in such drainage control programmes (e.g. the Interreg project) is encouraged by means of positive incentives (a restitution of taxes). Under such programmes, educational activities are started to convince farmers of the advantages of fine-tuned drainage control. Less drainage leads, for instance, to a better utilisation of minerals by the crop, which encourages the adoption of sophisticated drainage management tools, e.g. adjustable weirs. Voluntary agreements concerning drainage control seem to be very suitable since they involve the participants' specialised knowledge of local conditions. They are only effective, however, if such agreements are really established. The size of the transaction costs depends on the kind of agreement reached. When costs and benefits are not equitably distributed among the affected parties, both parties can bargain about compensation payments. The allocation of such payments depends on the assignment of rights. The acceptability is no problem as long as participation is encouraged by positive rather than negative incentives. Because of all these advantages, voluntary agreements for drainage control are highly recommended. The participation of farmers in decision-making at the local level is now becoming more common. The principle of allowing the individual members of agricultural organisations and water boards to make decisions on issues that affect them rather than leaving those decisions to be made by the whole group (the so-called principle of subsidiarity) is widely accepted.

In conclusion, Table 6.1 shows that most instruments are more or less effective as long as they are properly applied. A subsidy and a standard, however, do not achieve the objectives in an efficient way, while the introduction of a tax and a standard score low on acceptability. The suitability of a tax and tradable rights depends on the size of the efficiency gains relative to the transaction costs. However, the transaction costs of economic instruments for groundwater level management seem to be relatively high due to all kinds of practical implementation problems. Firstly, it is hard to define a good tax base. Secondly, there is heterogeneity in the vulnerability of areas to changes in drainage. Finally, the internalisation of externalities in the price of groundwater levels is very complicated, if not impossible, because it is hard to exclude farmers from changes in groundwater level management and because users do often not directly pay for using *in situ* services. Groups of farmers will be affected by changes in groundwater level. This implies that a change in the behaviour of one farmer with respect to groundwater level management, in response to an economic incentive, may affect adjacent farmers. Due to these interdependencies, third-party effects may arise. Changes in the institutional environment and voluntary agreements between groups of farmers and nature organisations seem to be promising for groundwater level management.

6.4 Suitability of policy instruments for groundwater extraction management

In this section the suitability of instruments for groundwater extraction management will be tested in the same way as instruments for groundwater level management were tested in Section 6.3. Table 6.2 shows results of instruments for groundwater extraction management.

Table 6.2 Suitability of instruments for groundwater extraction management(see Table 6.1 for the explanation of the '+' symbol for the various performance criteria).

Instrument	Effective-ness	Economic efficiency	Technical efficiency	Transaction costs	Equity	Acceptability
2A Institutional change	+	+	+	+/-	-	-
2B Tax	+/-	+	+	+/-	+/-	-
2C Tradable rights	+	+	+	+/-	+/-	+/-
2D Subsidy	+/-	-	-	+/-	-	+
2E Standard	+	-	-	+/-	+/-	-
2F Agreements	+/-	+	+	+/-	+	+/-

2A) *Institutional change* in the assignment of extraction rights seems interesting to reduce extraction in an efficient way. The current extraction rights system is based on free extraction permits granted by local authorities (provinces) in the past. These permits can be considered as historical extraction rights ('grandfathering rules'). The system only refused an extraction right if the proposed extraction could damage other users (Perdok and Wessel, 1998); damage to ecosystems was not taken into account until very recently. Nature is jeopardised under such a system. The current groundwater extraction rights system is not efficient, since current allocation rules are based on a 'queuing' system that restricts the trading of rights. It 'queues' in particular the users of *in situ* services, like nature.
Changes in the assignment of the extraction rights can reduce extraction in an efficient way in areas sensitive to desiccation. A restricted number of groundwater extraction rights have to be redistributed among farmers with the intention of allocating rights in an efficient and

equitable way, something which is not easy to establish. Differences in the vulnerability of ecosystems to extraction should be taken into account when extraction rights are assigned. Changes in rights are not so easy to adjust. Besides, the involved parties are generally very sensitive to changes in the rights system and it may therefore encounter serious resistance. The transaction costs depend on how easy it will be to implement such changes in practice.

2B) Extraction is currently regulated by two acts:
-Under the 1995 *Taxes on an Environmental Basis Act*, extraction is subject to a *tax*, which has to be paid to the central government. It is not a regulating tax, since the main aim is not to reduce water use, but to generate revenues. There are two tariffs: € 0.15/m^3 for waterworks and € 0.08/m^3 for other extractors. There is a tax-free threshold of 40,000 m^3/yr for extractions that are used for sprinkling. Extractions with pumps with a capacity of less than 10 m^3 an hour and extractions returned to the resource are also exempt from the tax. As extractions of waterworks are not exempt from this tax, the price of tap water has increased. This provided farmers with the incentive to sink their own wells, which means that such a tax is not very effective. On top of that, such diffuse extractions affect groundwater quality. We should therefore be careful with the creation of exceptions.
- Under the 1983 *Groundwater Act,* extraction is subject to a *levy*, which has to be paid to provinces. The tariffs and the levy-free threshold vary among provinces due to local differences and are subject to change. Tariffs are relatively small, for instance *€* 0.0136 per m^3 in the province of Noord-Brabant in 2001.

Currently only a small percentage of farmers (about 2%) exceed the tax-free threshold and are therefore subject to the tax (van Staalduinen et al. 1996). The main part of agricultural extraction is also not subject to the levy under the *Groundwater Act*. Most farmers currently only pay the energy costs of lifting water from the stock to the field (which is about € 0.04/m^3). The current price of water implies that externalities, which arise due to agricultural groundwater extraction, are not yet fully internalised in the price of water. This is not efficient, since farmers maximise individual (instead of social) current (instead of future) profit and pump water until its marginal net benefit is zero (in the absence of bans).

An alternative is to impose a higher tax, which can be considered as a kind of water pricing reform. Whether such a tax is justified depends on the allocation of property rights for extraction. It is, however, not clear who owns currently these property rights. The theoretical framework of the optimal tax level is clear (the Pigouvian tax on the activities of the generator of an externality has to be equal to the marginal externality cost produced by that activity), but difficulties emerge if a proper tax level has to be determined in reality. In case of local differences in the availability and quality of groundwater and the role groundwater plays with respect to terrestrial ecosystems, a differentiated tax system has to be created. The effectiveness depends on the risk attitude of farmers and price elasticity of water demand. This price elasticity seems to be too low to do an effective job, because even now sprinkling is not always economically profitable. Other aspects, like peace of mind, also play a role.

A tax is easy to adjust and increases flexibility. It reduces extraction where it is most efficient and improves the technical efficiency, since it raises the water price. The financial impact depends on the restitution of revenues. Usually, there is serious resistance to the introduction of a tax and transaction costs will be high. The abolition of the tax-free threshold of agricultural groundwater extraction under the *Taxes on an Environmental Basis Act* seems, however, to be promising for price reform against low transaction costs.

2C) *Tradable rights* are a restricted number of agricultural extraction rights that, once initially allocated by authorities, can be traded subject to a set of prescribed rules. Currently the transfer of rights to extract groundwater is already possible in the province of Brabant. Transition to agricultural water markets while diverting water away from agricultural use may decrease the agricultural sector's well-being to some extent, but is desirable from a social point of view (Shah and Zilberman, 1994). Tradable rights improve economic and technical efficiency, since the market determines the price of the right in a dynamic way. The high transaction costs are a disadvantage of water markets. An equitable introduction of water markets is, however, hard to establish. Rights can for instance be auctioned off, so that the authorities receive all the rent from new entitlements. An alternative is to allow senior rights owners to sell their water to buyers and benefit from the revenue so obtained. The financial impact on affected parties is determining for the acceptability. The use of tradable rights for groundwater extraction seems to be complicated in practice, since the impact of extraction on the desiccation of nature depends on location-specific circumstances. To avoid the transfer of extraction rights to regions sensitive to desiccation, guided trading is required. As the market will not take differences in the vulnerability of reserves to desiccation into account, it is necessary to intervene in the market to safeguard the environmental targets. The price of extraction rights is closely related to the heterogeneity criterion of water, geographic area, characteristics of the local market, size of the transaction, number and size of potential traders and the information and searching costs involved (Colby et al., 1993).

2D) A *subsidy* can be provided for meeting a certain volume of extraction, which is smaller than the standard. However, a subsidy scheme may make it profitable for a farmer to start off by extracting more than otherwise in order to qualify for larger subsidy payments. Subsidies fail to give a clear sign of real scarcity to farmers. They may be disturbing and do not provide incentives to adopt modern technologies. The acceptability is no problem, since participation in subsidy schemes is voluntary and because of the financial implications.

2E) Sprinkling bans (*standards*) currently aim to reduce low-value agricultural groundwater extraction. They divert water away from current agricultural use to non-agricultural and/or future use. These bans differ per province and vary with respect to the source of water used (groundwater versus surface water), crop type (grass versus arable), soil type (sandy versus clay) and time period (part of the year and day). Bans especially aim to reduce groundwater use for the sprinkling of grass on sandy soils in areas sensitive to desiccation during periods of drought. Farmers are not compensated financially for income losses due to sprinkling bans, which means that the extraction rights are not implicitly ascribed to agriculture. Current bans are only rough restrictions. For instance, there are no arable-crop-specific bans. Rough bans are suitable for a quick reduction in extraction and to ban a certain extractor.

An alternative is to fine-tune bans to resource-, region-, soil-, crop- and time-specific circumstances in such a way that they will allocate water efficiently. Bans provide incentives to change farming practices (e.g. the cropping pattern), but do not provide incentives to adopt modern technologies. The financial impact is not always equitably distributed among affected parties, since there are differences in the vulnerability of areas to extraction (and therefore in bans). Differentiated bans will increase the transaction costs considerably. Usually serious resistance is raised against the introduction of bans.

2F) *Voluntary agreements* currently induce participation in irrigation scheduling programmes. A management tool (often referred to as the sprinkling planner) for irrigation scheduling was developed and tested in 1995. Irrigation scheduling gives farmers a better insight into the moisture regime of their plots, the best time to start irrigation and the best water dose and prevents over-irrigation and thus increases the irrigation effectiveness. It is not likely that farmers will adopt the sprinkling planner under the low water prices they currently face. Motives for not adopting it are the investment costs it entails, its complexity and the effort its use will require. Practical test results showed that indeed only a small group of farmers would adopt the sprinkling planner of their own accord (Boland et al., 1996). Farmers are therefore subsidised and education and persuasion activities were started to induce participation (carrot approach). Persuasion is hard since farmers often behave myopically due to the competition in the sector. Besides, there will be no sprinkling bans if a certain diffusion rate is met (stick approach), i.e. if a certain number of farmers adopt the planner within a certain time period. Those who do not participate also benefit from the absence of bans and can be considered ‘free riders’. Another very promising voluntary agreement we recommend is a commitment between farmers and nature organisations concerning the extraction of groundwater. Voluntary agreements are, however, difficult to establish if rights remain poorly defined and are in that case not very effective. Voluntary agreements reduce extraction in an efficient and equitable way. The size of the transaction costs depends on the kind of agreement. The acceptability is often no problem.

In conclusion, Table 6.2 shows that most instruments are more or less effective in reducing groundwater extraction if they are properly applied. However, a subsidy and a standard do not achieve the objectives in an efficient way, while a change in the assignment of groundwater extraction rights and the introduction of a tax and a standard score low on acceptability. The suitability of a tax and tradable rights depends on the size of the efficiency gains relative to the transaction costs. The size of the transaction costs depends among others on the costs of registration of the large number of small-scale diffuse and irregular agricultural groundwater extractions. Voluntary agreements also seem to be promising.

6.5 Conclusions

As policy instruments have major impacts on the efficiency and effectiveness of water allocation and the adoption of modern technologies, we studied the suitability of policy instruments for groundwater level and extraction management. However, unambiguous statements about the suitability of policy instruments are hard to make without local empirical analysis, since the suitability differs locally due to region-specific circumstances.

Our theoretical analysis shows that the use of economic instruments for groundwater level management will be complicated in practice for several reasons. Changes in the institutional environment and voluntary agreements seem to be more suitable. It is for instance recommended to make water boards responsible for groundwater level management. Voluntary agreements between groups of farmers and nature organisations also seem to be promising. Such agreements are, however, difficult to establish if rights are poorly defined.

The current historical groundwater extraction rights system together with the low groundwater prices encourage low-value agricultural groundwater usage, whereas sprinkling bans and irrigation scheduling currently aim to reduce low-value use of groundwater. These extraction instruments are less efficient than a system that considers externalities in the price of water or diverts water away from agriculture while encouraging trading. Economic instruments seem therefore very promising for groundwater extraction management. They provide the correct incentives to use water more efficiently. In practice policy reform is usually conditional upon the size of efficiency gains relative to the transaction costs.

The distribution of transaction costs between agriculture and nature affects the optimal agricultural groundwater extraction reduction level. The larger the share of transaction costs agriculture has to bear; the smaller the reduction. The distribution of transaction costs among farms affects the efficiency gains of a transition from an extraction quota towards a tax. Gains increase if the divergence between the slopes of the marginal cost curves increases.

7 Summary and conclusions

7.1 Introduction

In the Netherlands, nature is largely adapted to wet and moist environmental conditions because of the country's low lying soil surface in combination with shallow groundwater levels. However, during the second half of the twentieth century, groundwater levels decreased, mainly due to the intensified drainage of agricultural areas and steadily increasing extraction of groundwater. As a result of these declining groundwater levels, nature in the Netherlands is suffering from desiccation.

The Dutch government aims to reduce the area of desiccated land by at least 40% by the year 2010 compared to the situation in 1985, when 600,000 hectares were suffering from desiccation. Since measures taken to raise groundwater levels in order to restore nature often lead to unintended wet damage to crops in adjacent farmland, economic analyses are required to determine optimal solutions.

The fundamental factors involved in such an analysis are presented in this thesis. The main objectives are: (1) to gain an economic insight into conflicting interests between agriculture and nature with respect to the desiccation of nature; (2) to develop methods and models to analyse groundwater level management; (3) to study agricultural groundwater extraction; and (4) to provide an insight into the suitability of policy instruments for both groundwater level and groundwater extraction management. This resulted in four main research questions that were formulated in Chapter 1 and will be answered in Section 7.2-7.5 respectively, summarising the main results. Section 7.6 contains recommendations for future research.

7.2 Economic and institutional causes of the desiccation of nature

Research question 1:
What are the economic and institutional causes of desiccation of nature in the Netherlands?

The study shows in the Chapters 2 and 6 that the failure of markets, institutions and policies has resulted in the desiccation of nature in the Netherlands.

Markets fail due to the public good nature of groundwater services and to externalities of groundwater extraction. It is hard to consider losses of nature as a result of drainage and agricultural groundwater extraction in the decision-making process about groundwater management for two reasons. Firstly, because of a lack of insight into the impact of hydrological changes on conservation values, measured by means of an ecological indicator. Secondly, water for nature is not marketed, and hence, not easily valued in monetary terms. Farmers receive therefore no signal through the market to take losses to nature into account.

Desiccation of nature is also the result of various *institutional failures*. Water boards make decisions about surface water levels, which directly affect groundwater levels, whereas they are currently not responsible for groundwater level management. Various interests with respect to this management are not well-balanced, since in the past within the governing body of the water boards mainly agricultural interests dominated. Besides, institutional settings are not always well-defined. It is for instance not yet clear who owns rights to lower groundwater levels and to extract groundwater. Since farmers are currently compensated financially for income losses due to higher groundwater levels, the rights to lower groundwater levels have so far been ascribed to agriculture. In the absence of institutional constraints, groundwater would be a *common resource*. In the Netherlands agricultural groundwater extraction is, however, currently based on *historical extraction rights*. Local authorities granted extraction rights, and only refused a right when the proposed extraction would damage other users, whereas damage to nature was not considered. Nature is jeopardised under such a system.

Policies can fail or function at the margin for various reasons. Firstly, if externalities are not internalised in policies, such as in the current price of agricultural groundwater extraction. Secondly, if policies cause third-party effects. Thirdly, policies may not be very effective if they are not properly implemented or exemptions are created, such as the tax-free threshold under the *Taxes on an Environmental Basis Act*. Finally, current policies fail if more efficient policies can be implemented against low transaction costs.

7.3 Optimal groundwater level management

Research question 2:
How can optimal groundwater level management be analysed?

The socially optimal groundwater level in agricultural areas with specific ecological value can be obtained by maximisation of the sum of the agricultural production value and the monetary value of ecological benefits (Chapter 3). At this optimum, the groundwater level is raised to the point where marginal costs of sacrificed production losses to agriculture equal marginal benefits to nature. It is, however, hard to find the monetary value of ecological benefits. In the cost-benefit analysis, several monetary values of ecological benefits have been studied, which are based on other studies. These values have only been used to illustrate the analysis. The study shows that the groundwater level will be more attuned to nature if a higher monetary value is attached to nature, if the share of agricultural area with ecological value in an entire agricultural area becomes larger and if the vegetation to restore is dominated by 'wet' species. The study also shows that higher groundwater levels will cause less wet damage to grass than to arable crops.

As hydrological restoration measures taken to raise the groundwater level often cause different levels of unintended wet damage to adjacent farmland, there is a need to gain insight into differences in cost-effectiveness of restoration. A multidisciplinary economic-hydrological model has been developed (Chapter 4). It provides insight into: (1) differences in costs to agriculture due to restoration per unit benefits to nature; (2) the possible division of payments to compensate costs to agriculture due to restoration; and (3) the cost-effectiveness of investments in hydrological measures to re-wet nature. Cost-effectiveness analysis instead of cost-benefit analysis is used, because no reliable monetary estimates of such benefits were available for the case study areas and benefit transfer is questionable.

Different objective functions have been formulated. They maximise (1) the total increase in the unaffected surface area; (2) the total increase in the ecological conservation value; and (3) the total utility of an increase in unaffected surface area. *It turns out that the current policy goal to reduce the desiccated surface area will not maximise the increase in the ecological conservation value.* It also appears that costs to agriculture due to restoration are small compared to investment costs of hydrological measures. Such investment costs vary considerably. Investment costs of the creation of hydrological buffer zones are for instance relatively high compared to investment costs of an increase in the groundwater recharge rate. The latter option is, however, neither always technically possible nor ecologically desirable.

In these analyses a trade-off between costs and benefits of restoration has been made. Costs will, however, only be justified if a continuation of measures is guaranteed over a long period, because otherwise their ecological benefits will not be realised. This necessary continuation of measures can be guaranteed by means of long-term policies.

7.4 Agricultural groundwater extraction management

Research question 3:
How can agricultural groundwater extraction management be analysed?

Groundwater extraction from agricultural areas may lead to desiccation of adjacent nature reserves and degradation of groundwater quality. Both externalities are, however, often not considered when agricultural groundwater extraction paths are being determined and are not considered in the price of groundwater. Optimal control theory has been used to study the dynamics of agricultural groundwater extraction management (Chapter 5). In the analysis the sum of discounted agricultural benefits and environmental damage is maximised over an infinite time horizon, taking into account shadow prices of changes in quantity and quality of groundwater over time.

The analysis shows that the current price of groundwater is too low and provides fewer incentives for the adoption of modern sprinkling techniques than the price under a system that internalises costs of desiccation and groundwater contamination. If these externalities are not internalised in the price of groundwater, contamination will be accelerated and quality improvements will be slowed down. This will affect the time path of changes in groundwater stock quality over time. This stresses the importance of considering groundwater quality when designing water-price reform, which is a key element of the European Water Framework Directive. Internalisation of externalities from agricultural shallow groundwater extraction on groundwater quality in the price of groundwater will be particularly significant, if the recharge rate of groundwater is large compared to stock size.

7.5 Policy recommendations

Research question 4:
What kind of policy reforms for groundwater management can be recommended?

The suitability of policy instruments for groundwater level and extraction management in the Netherlands has been reviewed against a number of performance criteria (Chapter 6).

As the suitability depends on location specific circumstances, unambiguous statements about the suitability of policy instruments are hard to make without full empirical analysis. Our theoretical analysis shows, nevertheless, that *the use of economic instruments for groundwater level management will be complicated in practice* for a number of reasons. Firstly, it is hard to define a good tax base. It is for instance not easy to measure the amount of drainage water. A tax on a change in the groundwater level is also complicated, because external factors also affect the groundwater level. Secondly, there is heterogeneity in vulnerability of areas to changes in drainage. Such region-specific circumstances require differentiated tax levels and guided trading of rights to lower groundwater levels. Finally, internalisation of externalities in the price of groundwater levels is very complicated, if not impossible, because it is usually hard to exclude farmers from changes in groundwater level management and because users do often not directly pay for using *in situ* services. This implies that the decision of one farmer to raise the groundwater level, in response to a tax, may affect adjacent farmers. Due to these interdependencies, third-party effects may arise.

Voluntary agreements between groups of farmers and nature organisations about drainage control seem to be more promising for *groundwater level management than economic instruments*, since it involves the participants' specialised knowledge of local conditions. It is based on the so-called principle of subsidiarity, which involves individuals in decisions on issues that affect them. Voluntary agreements are, however, difficult to establish if it is not well-defined who owns rights to lower groundwater levels. In that case negotiations about compensations payments under such agreements will be hard. *Institutional changes* seem therefore to be promising. It is for instance recommended to make water boards responsible for groundwater level management, so that conflicting interests with respect to groundwater level management can be better balanced within the governing body of water boards. Water boards may play an important role to prevent unnecessary damage and to repair damage against low costs by means of fine-tuned management taking account of various interests.

Economic instruments are suitable for *groundwater extraction management,* since they may influence individual farmers' behaviour by providing incentives to use water efficiently. A system that considers externalities in the price of water by means of tax or diverts water away from agriculture by means of tradable quota rights is for instance more efficient than current instruments that aim to reduce agricultural groundwater extraction, such as sprinkling bans and irrigation scheduling. This does, however, not automatically mean that economic instruments should be used in practice, as is often believed. If the efficiency gains of a transition from sprinkling bans towards higher groundwater prices are for instance relative small compared with additional transaction costs, sprinkling bans may be still preferable.

It is recommended to base decisions about *the restoration of nature reserves* on the cost-effectiveness of restoration and to maximise the increase in the ecological conservation value instead of the reduction in the desiccated surface area. Since marginal costs to agriculture per conservation value unit restored increase as more nature is restored, partial restoration of many reserves rather than the full restoration of only a few reserves could be considered.

7.6 Research recommendations

In this study a number of gaps in knowledge was revealed. Only those gaps are discussed, which are considered most relevant for future research.

• Currently there is a lack of insight into the magnitude of *transaction costs* of instruments for groundwater management or what factors influence the transaction costs. Such insight is, however, desirable, since policy reform is usually conditional upon the size of efficiency gains relative to transaction costs. The size of transaction costs depends among others on the costs of registration of the large number of small-scale diffuse and irregular agricultural groundwater extractions. The distribution of transaction costs determines the reduction in groundwater extraction of each party. *It is therefore recommended for future research to study the magnitude and distribution of transaction costs of groundwater policy reforms.*

• As the marginal value ascribed to nature will change with the total size of nature in a country, the marginal value of nature has to be endogenous in the analysis in case policy measures affect the size of nature reserves. In this study the marginal value of nature has, however, been kept constant, except in the utility function approach. A utility function approach is applied with a *declining marginal value of nature*, but the values of the weights of the utility function are difficult to establish. There is a lack of insight into the preferences of the public and policy makers for ecosystem types, as well as into the factors that influence these preferences, such as rarity. *It is recommended to model the marginal value of nature as an endogenous variable, and to test in an empirical setting preferences for ecosystem types.*

• Measures taken to reduce desiccation of nature may *interact with other environmental themes*, like acidification and eutrophication. Sprinkling and less drainage lead for instance to a better utilisation of minerals by the crops, which reduces leaching problems. While changes in the cropping pattern or intensity of agricultural production due to Common Agricultural Policy reform may lead to a change in the evapotranspiration of crops. *It is therefore recommended for future research to study whether environmental themes affect each other and to what extent policy measures reinforce each other*.

• Finally, it is important to note that for making the right choices about groundwater management, improvements in the understanding of relationships between the hydrological, economic and institutional systems are essential. The multidisciplinary approach used in this thesis contributes to the understanding of these relationships and is necessary for a full comprehension of not only the technical, but also the economic and institutional causes of the desiccation problem. A multidisciplinary approach is also necessary to gain insight into site-specific impacts of hydrological changes on crop growth and nature development. For insight into the suitability of policy instruments, empirical analyses of local conditions are required. *A multidisciplinary site-specific empirical approach by hydrologists, ecologists and agricultural/environmental economists is therefore recommended for designing successful strategies to combat the desiccation of nature.*

Samenvatting en conclusies

Inleiding

De natuur in Nederland is grotendeels aangepast aan natte en vochtige omstandigheden door de lage ligging van het land in combinatie met het ondiepe grondwaterpeil. Gedurende de tweede helft van de twintigste eeuw zijn grondwaterpeilen echter gedaald, voornamelijk ten gevolge van intensievere ontwatering van landbouwgebieden en de gestaag toegenomen winning van grondwater. Deze peilverlagingen hebben geleid tot verdroging van de natuur in Nederland. De Nederlandse overheid heeft doelstellingen geformuleerd om de verdroging terug te dringen. Ten opzichte van 1985, toen 600.000 hectare verdroogd was, dient in het jaar 2010 het verdroogde areaal met tenminste 40% te zijn teruggedrongen. Aangezien verhoging van het waterpeil vaak leidt tot natschade aan aangrenzende landbouwgewassen, is een economische analyse nodig om optimale oplossingen te bepalen.

De fundamentele factoren voor een dergelijke analyse worden in dit proefschrift behandeld. De belangrijkste doelstellingen zijn: (1) het verkrijgen van economisch inzicht in de conflicterende belangen tussen landbouw en natuur met betrekking tot de verdroging van de natuur; (2) het ontwikkelen van methoden en modellen voor het beheer van grondwaterpeilen; (3) het bestuderen van grondwaterwinning door de landbouw; en (4) het bieden van inzicht in de geschiktheid van beleidsinstrumenten voor zowel het beheer van het grondwaterpeil als de grondwaterwinning door de landbouw. Dit leidde tot de vier onderzoeksvragen die in Hoofdstuk 1 werden geformuleerd. Deze onderzoeksvragen worden hier beantwoord aan de hand van de belangrijkste bevindingen uit dit proefschrift.

Economische en institutionele oorzaken van verdroging van de natuur

Onderzoeksvraag 1:
Wat zijn de economische en institutionele oorzaken van verdroging van de natuur in Nederland?

In de Hoofdstukken 2 en 6 van deze studie werd duidelijk dat een combinatie van falen van markten, instituties en het beleid heeft geleid tot verdroging van de natuur in Nederland.

Er is sprake van marktfalen ten gevolge van het publieke karakter van grondwater en externe effecten door grondwaterwinning. Om twee redenen is het moeilijk om schade aan de natuur mee te nemen in beslissingen ten aanzien van grondwaterbeheer. Ten eerste door een gebrek aan inzicht in de gevolgen van hydrologische veranderingen op natuurwaarden. Ten tweede is het lastig om de waarde van water voor de natuur in monetaire termen uit te drukken, omdat de prijs van natuur niet op een markt tot uitdrukking komt. De landbouw ontvangt dan ook geen prikkel via de markt om schade aan de natuur in beschouwing te nemen.

Verdroging van de natuur is ook het gevolg van het *falen van instituties*. Waterschappen nemen beslissingen ten aanzien van oppervlaktewaterbeheer, wat direct het grondwaterpeil beïnvloedt, terwijl ze op dit moment niet verantwoordelijk zijn voor het grondwaterbeheer. Lange tijd is het beheer voornamelijk afgestemd op de belangen van de agrarische sector. Bovendien zijn instituties niet altijd goed gedefinieerd. Het is bijvoorbeeld niet duidelijk wie de rechten bezit voor het verlagen van grondwaterpeilen en voor de grondwaterwinning. Aangezien de landbouw momenteel financieel gecompenseerd wordt voor inkomensverliezen ten gevolge van hogere grondwaterpeilen, lijkt het erop dat de rechten voor het verlagen van grondwaterpeilen toegeschreven kunnen worden aan de landbouw. Zonder institutionele beperkingen is grondwater een gemeenschappelijke hulpbron. In Nederland is de agrarische grondwaterwinning gebaseerd op historische vergunningen. Locale overheden verleenden deze vergunningen en weigerden alleen een vergunning als de voorgenomen winning schade aan anderen toebracht, waarbij schade aan de natuur niet in beschouwing werd genomen.

Om uiteenlopende redenen kan *het beleid slecht functioneren*. Ten eerste als externe effecten niet geïnternaliseerd zijn in het beleid, zoals in de prijs van agrarische grondwaterwinning. Ten tweede als beleid schade aan derde partijen toebrengt. Ten derde kan het beleid weinig doeltreffend zijn als het niet op de juiste manier is geïmplementeerd of uitzonderingen creëert, zoals de heffingsvrije zone onder *de Wet Belasting op Milieu-Grondslag*. Tenslotte faalt beleid als een efficiënter beleid kan worden geïntroduceerd tegen lage transactiekosten.

Optimaal beheer van het grondwaterpeil

Onderzoeksvraag 2:
Hoe kan optimaal beheer van het grondwaterpeil worden geanalyseerd?

Het sociaal optimale grondwaterpeil in agrarische gebieden met bijzondere ecologische waarden kan worden verkregen door het maximaliseren van de som van de agrarische productiewaarde en de monetaire waarde van ecologische baten van agrarisch natuurbeheer (Hoofdstuk 3). In dit optimum zijn de marginale kosten van de landbouw gelijk aan de marginale baten van de natuur. In de kosten-batenanalyse zijn uiteenlopende monetaire waarden voor ecologische baten uit diverse studies bestudeerd. Deze waarden zijn alleen gebruikt om de werking van het model te illustreren. De studie laat zien dat het grondwaterpeil beter wordt afgestemd op de natuur als een hogere monetaire waarde aan de natuur wordt toegekend, als het aandeel van landbouwgrond met ecologische waarde in het totale landbouwareaal toeneemt en als de te herstellen vegetatie vooral bestaat uit 'natte' soorten. De studie laat eveneens zien dat hogere grondwaterpeilen minder natschade toebrengen aan gras dan aan akkerbouwgewassen.

Aangezien hydrologische herstelmaatregelen vaak leiden tot verschillen in natschade aan aangrenzende landbouwgewassen, is inzicht in de kosteneffectiviteit van herstel wenselijk. Hiervoor is een multidisciplinair economisch-hydrologisch model ontwikkeld (Hoofdstuk 4). Het biedt inzicht in: (1) verschillen in kosten van de landbouw als gevolg van herstel per eenheid baten voor de natuur; (2) de mogelijke verdeling van betalingen ter compensatie van schade aan de landbouw door natuurherstel; en (3) de kosteneffectiviteit van investeringen in hydrologische maatregelen om de natuur te vernatten. Kosteneffectiviteit analyse is gebruikt in plaats van kosten-batenanalyse, omdat er geen betrouwbare monetaire schattingen van de baten beschikbaar waren voor de case studiegebieden en 'benefit transfer' dubieus is.

Diverse doelfuncties zijn geformuleerd. De doelfuncties maximaliseren (1) de afname in het verdroogde areaal; (2) de toename in de natuurwaarde gemeten op een ecologische schaal; en (3) het nut van een afname in het verdroogde areaal. *Hieruit blijkt dat de huidige beleidsdoelstelling om het aantal hectaren verdroogd areaal te reduceren, niet de toename in de natuurwaarde maximaliseert*. Het wordt ook duidelijk dat kosten van natschade voor de landbouw door herstel klein zijn ten opzichte van investeringskosten van hydrologische maatregelen. Investeringskosten variëren aanzienlijk. De kosten voor het creëren van bufferzones zijn bijvoorbeeld hoog ten opzichte van kosten van grondwateraanvulling. Deze laatste optie is echter niet altijd technisch mogelijk noch ecologisch wenselijk.

In deze analyses worden kosten en baten van herstel tegen elkaar afgewogen. Kosten zijn echter uitsluitend gerechtvaardigd indien maatregelen worden voortgezet over een langere tijdsperiode, omdat anders geen duurzaam ecologisch herstel wordt gerealiseerd. Deze noodzakelijke voortzetting kan worden gegarandeerd middels lange termijn beleid.

Beheer van grondwaterwinning door de landbouw

Onderzoeksvraag 3:
Hoe kan het beheer van agrarisch grondwaterwinning worden geanalyseerd?

Grondwaterwinning in landbouwgebieden kan leiden tot verdroging van aangrenzende natuurgebieden en verandering in de grondwaterkwaliteit. Beide externe effecten worden vaak buiten beschouwing gelaten bij beslissingen met betrekking tot grondwaterwinning en worden niet meegenomen in de prijs van grondwater. Optimal control theory is gebruikt om agrarische grondwaterwinning over langere tijdsperioden te bestuderen (Hoofdstuk 5). In de analyse wordt de som van de gedisconteerde agrarische baten en schade aan de natuur gemaximaliseerd over een oneindige tijdshorizon, rekening houdend met de schaduwprijzen van veranderingen in de kwantiteit en kwaliteit van grondwater in de loop van de tijd.

De analyse laat zien dat de huidige grondwaterprijs te laag is en minder prikkels geeft voor de omschakeling naar efficiëntere beregeningstechnieken dan de prijs onder een systeem dat kosten van verdroging en vervuiling in de prijs meeneemt. Als externe effecten niet worden geïnternaliseerd in de prijs wordt grondwatervervuiling versneld en kwaliteitsverbetering afgeremd. *Dit benadrukt het belang van het in beschouwing nemen van grondwaterkwaliteit bij hervormingen van de waterprijs, hetgeen een element is van de Europese Kaderrichtlijn Water.* Het internaliseren van effecten van grondwaterwinning op de grondwaterkwaliteit is van belang indien de grondwateraanvulling groot is ten opzichte van de grondwatervoorraad.

Beleidsaanbevelingen

Onderzoeksvraag 4:
Welk type beleidshervormingen voor grondwaterbeheer kan worden aanbevolen?

De geschiktheid van beleidsinstrumenten voor het beheer van het grondwaterpeil en het beheer van grondwaterwinning in Nederland is geëvalueerd aan de hand van een aantal toetsingscriteria (Hoofdstuk 6). Aangezien de geschiktheid afhangt van locatiespecifieke omstandigheden, is het moeilijk een eenduidige uitspraak te doen over de geschiktheid van beleidsinstrumenten voor grondwaterbeheer zonder empirische analyses. De theoretische analyse laat desalniettemin zien dat het gebruik van economische instrumenten voor het beheer van het grondwaterpeil om verschillende redenen gecompliceerd is in de praktijk. Ten eerste is het moeilijk om een grondslag voor een heffing op het beheer van het grondwaterpeil te definiëren. Het is bijvoorbeeld lastig om de hoeveelheid ontwatering te meten. Een heffing op een verandering in het grondwaterpeil is ook lastig, aangezien externe factoren het grondwaterpeil beïnvloeden. Ten tweede verschilt de kwetsbaarheid van gebieden ten aanzien van ontwatering. Zulke locatiespecifieke omstandigheden vragen om een gedifferentieerde heffing en gecontroleerde handel in rechten voor het verlagen van grondwaterpeilen. Tenslotte is het internaliseren van externe effecten ingewikkeld, omdat het vaak moeilijk is om landbouwgrond uit te sluiten van veranderingen in het grondwaterpeil en omdat gebruikers vaak niet direct betalen voor een bepaald grondwaterpeil. Dit impliceert dat de beslissing van een landbouwer om in reactie op een heffing het grondwaterpeil te verhogen, schade kan toebrengen aan aangrenzende landbouwers (derde partijen).

Vrijwillige overeenkomsten tussen de landbouw en natuurorganisaties lijken meer geschikt voor het beheer van grondwaterpeilen dan economische instrumenten, aangezien gebruik gemaakt wordt van specialistische kennis van de deelnemers over locale condities. Dit berust op het zogenaamde subsidiariteitsprincipe, waarbij individuen betrokken worden bij beslissingen die hen aangaan. Vrijwillige overeenkomsten zijn echter moeilijk tot stand te brengen indien niet duidelijk gedefinieerd is wie de rechten bezit voor het verlagen van grondwaterpeilen. Onderhandelingen over compensatiebetalingen zijn moeilijk in dat geval. Institutionele veranderingen lijken dan ook veelbelovend. Het verdient bijvoorbeeld aanbeveling om waterschappen verantwoordelijk te maken voor het operationele grondwaterbeheer, zodat conflicterende belangen met betrekking tot peilbeheer beter worden afgewogen. Waterschappen kunnen een belangrijke rol spelen in het voorkomen van onnodige schade en bij het herstel van schade tegen lage kosten door middel van een meer genuanceerd peilbeheer waarbij rekening gehouden wordt met de uiteenlopende belangen.

Economische instrumenten zijn geschikt voor het beheer van *grondwaterwinning*, aangezien ze het individuele gedrag van landbouwers kunnen beïnvloeden door het geven van prikkels om water efficiënter te gebruiken. Dit betekent echter niet dat economische instrumenten automatisch moeten worden gebruikt in de praktijk. Als bijvoorbeeld de efficiencywinst van een omschakeling van beregeningsverboden naar een heffing klein is ten opzichte van de hiermee samenhangende transactiekosten, kunnen beregeningsverboden wenselijk blijven.

Het verdient aanbeveling om beslissingen ten aanzien van het herstel van natuurgebieden te baseren op de kosteneffectiviteit van herstel en om de toename in de ecologische natuurwaarden in plaats van de afname van het verdroogde areaal te maximaliseren. Aangezien marginale kosten van de landbouw per eenheid natuurherstel toenemen als meer natuur is hersteld, kan gedeeltelijk herstel van veel natuurgebieden worden overwogen in plaats van volledig herstel van slechts enkele natuurgebieden.

Onderzoeksaanbevelingen

Tijdens dit onderzoek kwam een aantal hiaten in de kennis naar voren. Hier worden alleen de hiaten bespoken die voor toekomstig onderzoek het meest relevant worden geacht.

• Er is momenteel een gebrek aan inzicht in de omvang van de transactiekosten van de instrumenten voor grondwaterbeheer en in de factoren die de transactiekosten beïnvloeden. Dit inzicht is wenselijk, aangezien beleidshervormingen vaak afhangen van de omvang van de efficiencywinst ten opzichte van de transactiekosten. De omvang van de transactiekosten hangt onder andere af van de kosten van het registreren van het grote aantal kleinschalige, diffuse en onregelmatige agrarische winningen. *Het verdient aanbeveling om de omvang van transactiekosten van beleidshervormingen te bestuderen en kwantificeren.*

• Aangezien de marginale waarde die aan de natuur wordt toegeschreven afhangt van de totale omvang van de natuur in een land, moet de marginale waarde van de natuur binnen de analyse worden bepaald indien het geanalyseerde beleid de omvang van natuurgebieden beïnvloedt. In deze studie is de marginale waarde van de natuur echter constant gehouden, behalve in de doelfunctie die het nut van de afname in het verdroogde areaal maximaliseert. Een nutsfunctie met een afnemende marginale waarde van de natuur is gebruikt. De waarden van de gewichten van de nutsfunctie zijn echter moeilijk vast te stellen. Er is een gebrek aan inzicht in de preferenties van het publiek en van beleidsmakers voor ecosysteemtypen en in de factoren die deze preferenties beïnvloeden, zoals zeldzaamheid. *Het verdient dan ook aanbeveling om de marginale waarde van de natuur te modelleren als een endogene variabele en om in een empirische setting de preferenties voor ecosysteemtypen te testen.*

• Anti-verdroging maatregelen kunnen interveniëren met overig beleid of andere milieuthema's, zoals verzuring en eutrofiering. Beregening en minder ontwatering leiden bijvoorbeeld tot een betere opname van mineralen, waardoor uitspoeling vermindert. Terwijl veranderingen in de intensiteit van de agrarische productie of het bouwplan ten gevolge van hervorming van het Gemeenschappelijk Landbouw Beleid kunnen leiden tot een verandering in de evapotranspiratie van gewassen. *Het verdient dan ook aanbeveling om te bestuderen hoe milieuthema's elkaar beïnvloeden en in welke mate maatregelen elkaar versterken.*

• Tenslotte, is het belangrijk om op te merken dat voor het maken van de juiste keuzes met betrekking tot grondwaterbeheer, de relaties tussen het hydrologische, economische en institutionele systeem nader dienen te worden bestudeerd. De multidisciplinaire benadering zoals gebruikt in dit proefschrift, biedt niet alleen inzicht in de technische, economische en institutionele oorzaken van de verdroging, maar is ook van belang bij het bepalen van locatiespecifieke gevolgen van hydrologische veranderingen op natuurontwikkeling en gewasgroei. De bruikbaarheid van instrumenten kan het beste middels empirische analyses worden bestudeerd. *Voor het ontwerpen van succesvolle strategieën tegen de verdroging van de natuur verdient een multidisciplinaire locatiespecifieke empirische benadering door hydrologen, ecologen en agrarische milieu-economen dan ook aanbeveling.*

References

Adger W.N. and C. Luttrell, 2000, Property rights and the utilisation of wetlands. *Ecological Economics*, 35:75-89.

Albiac, J., S. Uku and Y. Martínez, 2001, *Water quality protection through irrigation management*. Paper presented at the seventh conference of the international water and resource economics consortium and fourth seminar on environmental and resource economics, 3-5 June, 2001, Girona.

Ancot, J.P. and T.J. van de Nes, 1981, *Integral evaluations of alternative water management scenarios in East-Gelderland*. TNO, Proceedings and Information No.27, The Hague.

Bhatia, R., R. Cestti and J. Winpenny, 1994, *Policies for water conservation and reallocation: good practice cases in improving efficiency and equity*. World Bank, Technical Paper, Washington.

Blumenthal, K.P. and H.A. Visée, 1988, Institutional aspects of the water resources system. In: R. Koudstaal, H.A. Pennekamp and J. Wesseling (eds.), *Planning for water resources management in The Netherlands*. Delft Hydraulics, Delft.

Boland, D., H. Bleumink and J.C. Buys, 1996, *Naar een beregeningsplanner voor agrariërs II, Toetsing in de praktijk* [Towards a sprinkling planner for farmers II, Testing it in practice]. Centrum voor Landbouw en Milieu, Report 235, Utrecht.

Boogerd, A., P. Groenewegen and M. Hisschemöller, 1997, Knowledge utilisation in water management in the Netherlands related to desiccation. *American Water Resources Association*, 3(4):731-740.

Briscoe, J., 1996, *Water as an economic good: the idea and what it means in practice*. Paper presented at the World Congress of the International Commission on Irrigation and Drainage, Cairo.

Bromley, D., 1991, *Environment and economy: property rights and public policy*. Basil Blackwell Ltd, Oxford.

Brouwer, R. and L.H.G. Slangen, 1998, Contingent valuation of the public benefits of agricultural wildlife management: the case of Dutch peat meadow land. *European Review of Agricultural Economics*, 25:53-72.

Bruggeman, G.A., 1999, *Analytical solutions to geo-hydrological problems*. Elsevier, Amsterdam.

Burness, H.S. and T.C. Brill, 2001, The role for policy in common pool groundwater use. *Resource and energy economics*, 23:19-40.

Burt, O.R., 1964, Optimal resource use over time with an application to groundwater. *Management Science*, 11(1):80-93.

Burt, O.R., 1975, Groundwater management and surface water development for irrigation. In: R.M. Thrall (ed.), *Economic modelling for water policy evaluation*, North Holland, Amsterdam.

Byström, O., 1998, The nitrogen abatement cost in wetlands. *Ecological Economics*, 26:321-331.

Caswell, M. and D. Zilberman, 1985, The choices of irrigation technologies in California. *American Journal of Agricultural Economics*, 67(2):224-234.

Caswell, M. and D. Zilberman, 1986, The effects of well depth and land quality on the choice of irrigation technology. *American Journal of Agricultural Economics*, 68(4):798-811.

Chiang, A.C., 1992, *Elements of dynamic optimisation*. McGraw-Hall Inc., London.

Coase, R., 1960, The problem of social cost. *Journal of Law and Economics*, 3:1-44.

Colby, B.G., K. Crandall and D.B. Bush, 1993, Water right transactions: market values and price dispersion. *Water Resources Research*, 29(6):1565-1572.

Colenbrander, H.J., K.P. Blumenthal, W. Cramer, A. Volker and J. Wesseling (eds.), 1989, *Water in the Netherlands*. Lakerveld bv, The Hague.

CVG (Committee on Valuing Groundwater), 1997*, Valuing groundwater: economic concepts and approaches*. National Research Council, National Academy Press, Washington.

Conrad, J.M. and C.W. Clark, 1987, *Natural resource economics: notes and problems*. Cambridge University Press, Cambridge.

Costanza, R., R. d'Arge, R.S. de Groot, S. Farber, M. Grasso, B. Hannon, K. Limburg, S. Naeem, R.V. O'Neill, J. Paruelo, R.G. Raskin, P. Sutton, and M. van den Belt, 1997, The value of the world's ecosystem services and natural capital. *Nature*, 387:253-260.

de Groot, R.S., 1992, *Functions of nature: evaluation of nature in environmental planning, management and decision making*. Wolters-Noordhoff, Groningen.

Dellink, R. and F. van der Woerd, 1997, *Kosteneffectiviteit van milieuthema's* [Cost-effectiveness of environmental themes]. Institute for Environmental Studies, R-97/10, Amsterdam.

Dijk, J., C. Ploeger and M.W. Hoogeveen, 1994, *Grondwateronttrekking door de land- en tuinbouw* [Groundwater extraction by agriculture and horticulture]. Agricultural Economics Research Institute, Publication 3.157, The Hague.

Dijkema, M.P., R.D.W. Hijdra, L. van der Meulen, J.P.M. Witte and G. van Wirdum, 1985, *Ecohydrologische beschrijvingen en vergelijking van een tiental natuurgebieden* [Ecohydrological description and comparison of nature reserves]. Studiecommissie Waterbeheer Natuur Bos en Landschap, Report 1b, Utrecht.

Dijkstra, B.R., 1998, *The political economy of instrument choice in environmental policy*. PhD thesis, University of Groningen, Groningen.

Dinar, A. and A. Wolf, 1994, International markets for water and the potential for regional co-operation: economic and political perspectives in the Western Middle East. *Economic Development and Cultural Change,* 43:44-66.

Dinar, A. and A. Xepapadeas, 1998, Regulating water quantity and quality in irrigated agriculture. *Journal of Environmental Management* 54:273-289.

Dinar, A. and D. Zilberman, 1991, Effects of input quality and environmental conditions on selection of irrigation technologies. In: A. Dinar and D. Zilberman (eds.), *The economics and management of water and drainage in agriculture*. Kluwer Academic Publishers, Boston.

Dosi, C. and K.W. Easter, 2000, *Water scarcity: economic approaches to improving management*. International Food and Agricultural Policy, WP00-2, St.Paul.

Dufour, F.C., 2000, *Groundwater in the Netherlands, facts and figures*. TNO, Delft/Utrecht.

FAO, 1994, *Reforming water resources policy: a guide to methods, processes and practices.* Food and Agriculture Organisation, Irrigation and Drainage Paper No.52, Rome.

Feddes, R.A., R.W.R. Koopmans and J .C. van Dam, 1999, *Agrohydrology*. Wageningen University, Department of Water Resources, Reader, Wageningen.

Feddes, R.A. and A.L.M. van Wijk, 1990, Dynamic land capability model: a case history. *Philosophical Transactions of the Royal Society of London, Series B,* 329:411-419.

Fleming, R.A. and R.M. Adams, 1997, The importance of site-specific information in the design of policies to control pollution. *Journal of Environmental Economics and Management,* 33:347-358.

Garrido, A., 1999, Pricing for water use in the agricultural sector. In: *Pricing water, economics, environment and society*. Conference Proceedings of DG Environment of the European Commission, 6-7 September, 1999, Sintra.

Gibbons, D.C., 1986, *The economic value of water: resources for the future*. Washington.

Gisser, M. and D.A. Sanchez, 1980, Competition versus optimal control in groundwater pumping. *Water Resources Research,* 16(4):638-642.

Groeneveld, R., A.F. Bouwman, S. Kruitwagen and E.C. van Ierland, 1998, Nitrate leaching in dairy farming: economic effects of environmental restrictions. *Environmental pollution,* 102(S1):755-761.

Grootjans, A.P., 1985. *Changes of groundwater regime in wet meadows*. PhD thesis, University of Groningen, Groningen.

GWP, 2000, *Integrated water resources management*. Technical Advisory Committee, Global Water Partnership, Paper No.4, Stockholm.

Hanley, N and C.L. Spash, 1993, *Cost-benefit analysis and the environment*. Edward Elgar, Aldershot.

Hartwick, J.M. and N.D. Olewiler, 1998, *The economics of natural resource use.* Addison-Wesley, Reading.

Heinz, I., 2001, Co-operative agreements to improve efficiency and effectiveness of policy targets. In: F.M. Brouwer and J. van der Straaten (eds.), *New perspectives on policies that shape the European Countryside.* Edward Elgar, Cheltenham.

Hodge, I. and S. McNally, 2000, Wetland restoration, collective action and the role of water management institutions. *Ecological Economics*, 35:107-118.

Howe, C.W., 1971, *Benefit-cost analysis for water system planning*. American Geophysical Union, Water Resources Monograph No.2, Washington.

Howitt, R.E. and J.R. Lund, 1999, Measuring the economic impacts of environmental reallocations of water in California. *American Journal of Agricultural Economics,* 81(5):1268-1272.

ICWE, 1992, *The Dublin statement and report of the International Conference on Water and the Environment.* Development issues for the 21st century, 26-31 January, 1992, Dublin.

Jansen., A.J.M, 2000, *Hydrology and restoration of wet heathland and fen meadow communities*. PhD thesis, Wageningen University, Wageningen.

Jansen, P.C. and H. Runhaar, 2000, Afwegingen bij het vaststellen van het optimale grondwaterregime voor natuurgebieden [Trade-offs to determine the optimal groundwater regime in nature reserves]. *Stromingen,* 6(1):23-31.

Kemmers, R.H., 1986, Perspectives in modeling of processes in the root zone of spontaneous vegetation at wet and damp sites in relation to regional water management. In: J.C. Hooghart (ed.), *Water management in relation to nature, forestry and landscape management*. TNO Committee on Hydrological Research No.34, The Hague.

Klijn, F., 1997, *A hierarchical approach to ecosystems and its applications for ecological land classification: with examples of eco-regions, eco-districts and eco-series of the Netherlands*. PhD thesis, Leiden University, Leiden.

Klijn, F., C.L.G. Groen and J.P.M. Witte, 1996, Ecoseries for potential site mapping, an example from the Netherlands. *Landscape and Urban Planning*, 35:53-70.

Klijn, F., J. Runhaar and M. van ’t Zelfde, 1997, *Ecoseries-2.1: verbetering en operationalisatie van een classsificatie van ecoseries voor DEMNAT-2.1* [Ecoseries 2.1- a classification of ecoseries for DEMNAT-2.1]. Technical report, RIZA, Lelystad.

Kneese, A.V., 1995, *Natural resource economics: selected papers of Allen V. Kneese*. Edward Elgar, Aldershot.

Kruitwagen, S., H. Folmer, E. Hendrix, L. Hordijk and E.C. van Ierland, 2000, Trading sulphur emissions in Europe: ‘Guided bilateral trade’, *Environmental and Resource Economics*, 16:423-441.

Kuks, S.M.M., 1998, Policy framework –The Netherlands. In: G.J.I. Schrama (ed.), *Drinking water supply and agricultural pollution preventive action by the water supply sector in the European Union and the United State*s. Kluwer Academic Publishers, Dordrecht.

Larson, D.M., G.E. Helfand and B.W. House, 1996, Second-best tax policies to reduce non-point source pollution. *American Journal of Agricultural Economics,* 78:1108-1117.

Lee, D.J. and R.E. Howitt, 1996, Modelling regional agricultural production and salinity control alternatives for water quality policy analysis. *American Journal of Agricultural Economics*, 78:41-53.

LEI, various years, Dutch Farm Accountancy Data Network (FADN). Agricultural Economics Research Institute, The Hague.

Londo, M., L. Vleeshouwer, J. Dekker and H. de Graaf, 2000, Energy farming in Dutch desiccation abatement areas: yields and benefits compared to grass cultivation. In: E.C. van Ierland, A. Oude Lansink and E.C. Schmieman (eds.), *Sustainable Energy: new challenges for agriculture and implications for land use.* Conference proceedings, Wageningen University, Wageningen.

Loomis, J., M. Hanemann, B. Kanninen and T. Wegge, 1991, Willingness to pay to protect wetlands and reduce wildlife contamination from agricultural drainage. In: A. Dinar and D. Zilberman (eds.), *The economics and management of water and drainage in agriculture*. Kluwer Academic Publisher, Boston.

Meeusen, M.J.G., M.H. Hoogeveen and H.C. Visee, 2000, *Waterverbruik in de Nederlandse land- en tuinbouw in 1997* [Water use by Dutch agriculture and horticulture in 1997]. Agricultural Economics Research Institute, Report 2.00.02, The Hague.

Michelsen, A.M. and R.A. Young, 1993, Optioning agricultural water rights for urban water supplies during drought. *American Journal of Agricultural Economics*, 75:1010-1020.

Musgrave, R.A. and P.B. Musgrave, 1976, *Public finance in theory and practice*. McGraw-Hill, New York.

OECD, 1989, *Water Resource Management: integrated policies*. OECD, Paris.

OECD, 1991, *Environmental policy: how to apply economic instruments.* OECD, Paris.

OECD, 2000, *Environmental performance reviews: Hungary*. OECD, Paris.

Oltmer, K., 1999, *Economic instruments in water management and desiccation in the Netherlands*. Agricultural Economics Research Institute, Report 2.99.05, The Hague.

Palma, C.R., 1999, *Groundwater management when water quality is endogenous*. Paper presented at the ninth annual conference of the EAERE, 25-27 June, 1999, Oslo.

Pearce, D. and D. Moran, 1994, *The economic value of biodiversity*. Earthscan Publications Ltd, London.

Pearce, D. and R. Turner, 1990, *Economics of natural resources and the environment*. Harvester Wheatsheaf, Hertfordshire.

Perdok, P.J. and J. Wessel, 1998, Netherlands. In: F.N. Correia (ed.), *Institutions for water resources management in Europe.* Balkema, Rotterdam.

Perman, R., Y. Ma, J. McGilvray and M. Common, 1999, *Natural resource and environmental economics*. Longman, Harlow.

Pindyck, R.S. and D.L. Rubinfeld, 1989, *Microeconomics*. Mac Millan, New York.

Provencher, B. and O. Burt, 1994, A private property rights regime for the commons: the case of groundwater. *American Journal of Agricultural Economics,* 76:875-888.

Randall, A., 1981, *Property entitlements and pricing policies for a maturing water economy. The Australian Journal of Agricultural Economics*, 25(3):195-220.

Renzetti, S., 2000, An empirical perspective on water pricing reforms. In: A. Dinar (ed.), *The political economy of water pricing reforms.* Oxford University Press, New York.

Rogers, P., R. Bhatia and A. Huber, 1998, *Water as a social and economic good: how to put the principle into practice.* Technical Advisory Committee, Global Water Partnership, Paper No.2, Stockholm.

Rosegrant, M.W., 1997, *Water resources in the twenty-first century: challenges and implications for action.* International Food Policy Research Institute, Discussion Paper No.20, Washington.

Runhaar, J., 1999, *Impact of hydrological changes on nature conservation in the Netherlands*. PhD thesis, Wageningen University, Wageningen.

Runhaar, J., C. Maas, A.F.M. Meuleman and L.M.L. Zonneveld, 2000, *Herstel van natte en vochtige ecosystemen: handboek* [Restoration of wet and moist ecosystems: handbook]. Nationaal Onderzoeksprogramma Verdroging, Report No.9-2, Lelystad.

Savenije, H., 2000, *The socio-economic value of water: knowledge gaps and research questions in integrated water resources management.* Paper presented at the UNESCO-WOTRO International Working Conference on Water for Society, 8-10 November, 2000, Delft.

Schmieman, E.C., 2001, *Acidification and tropospheric ozone in Europe: towards a dynamic economic analysis*. PhD thesis, Wageningen University, Wageningen.

Schmieman, E.C. and E.C. van Ierland, 1999, Dynamics of soil acidification: an economic analysis. *Ecological Economics*, 31:449-462.

Segerson, K. and T.J. Miceli, 1998, Voluntary environmental agreements: good or bad news for environmental protection? *Journal of environmental economics and management,* 36:109-130.

Shah, F. and D. Zilberman, 1991, Government policies to improve inter-temporal allocation of water in regions with drainage problems. In: A. Dinar and D. Zilberman (eds.), *The economics and management of water and drainage in agriculture*. Kluwer Academic Publishers, Boston.

Shah, F. and D. Zilberman, 1994, *Queuing vs. Markets in Water Resource Allocation*. Department of Agricultural and Resource Economics, University of California, Working Paper, Berkeley.

Shah, F., D. Zilberman and U. Chakravorty, 1995a, Technology adoption in the presence of an exhaustible resource: the case of groundwater extraction. *American Journal of Agricultural Economics*, 77:291-299.

Shah, F., D. Zilberman and E. Lichtenberg, 1995b, Optimal combination of pollution prevention and abatement policies: the case of agricultural drainage. *Environmental and Resource Economics*, 5:29-49.

Slangen, L.H.G., 1992, Policies for nature and landscape conservation in Dutch agriculture: an evaluation of objectives, means, effects and programme costs. *European Review of Agricultural Economics*, 19:331-350.

Silvis, H.J. and C. van Bruchem (eds.), 2000, *Landbouw-Economisch Bericht 2000*. Agricultural Economics Research Institute, PR 1-00, The Hague.

Sinha, B. and R. Bhatia, 1982, *Economic appraisal of irrigation projects in India*. Studies in economic development and planning No.27 Agricole Publishing Academy, New Delhi.

Spash, C.L. and C. Carter, 2001, Environmental valuation methods in rural resource management. In: F.M. Brouwer and J. van der Straaten (eds.), *New perspectives on policies that shape the European Countryside*. Edward Elgar, Cheltenham.

Spieksma, J.F.M., 1998, *Rewetting of damaged peatlands*. PhD thesis, Wageningen University, Wageningen.

Stevenson, G.G., 1991, *Common property economics: a general theory and land use application*. Cambridge University Press, Cambridge.

Strosser, P., 1997, *Analysing alternative policy instruments for the irrigation sector: an assessment of the potential for water market development in the Chishtian Sub-division.* PhD thesis, Wageningen University, Wageningen.

SWWF, 2000, *Forum report on water and economics of the Second World Water Forum*, The Hague.

Turner, R.K. and J.B. Opschoor, 1994, Environmental economics and environmental policy instruments: introduction and overview. In: J.B. Opschoor and R.K. Turner (eds.), *Economic incentives and environmental policies: principles and practice.* Kluwer Academic Publishers, Dordrecht.

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

UNCED, 1992, *United Nations Conference on Environment and Development, Final communication*. Rio de Janeiro.

Union of Water boards, 1999, *Water boards*. Unie van Waterschappen, The Hague.

van Amstel, A.R., L.C. Braat, A.C. Garritsen, C.R. van Gool, N. Gremmen, C.L.G. Groen, H.L.M. Rolf, J. Runhaar and J. Wierz, 1989. *Verdroging van natuur en landschap in Nederland: beschrijving en analyse* [Desiccation of nature and landscapes in the Netherlands]. Ministry of Transport and Public Works, Technical report, The Hague.

van Beusekom, C.F., J.M.J. Farjon, F. Foekema, B. Lammers, J.G. de Molenaar and W.P.C. Zeeman, 1990, *Handboek grondwaterbeheer voor natuur, bos en landschap* [Handbook groundwater management for nature, wood and landscape]. SDU, The Hague.

van Buuren, M., 1997, *Landschapsplanning en watersystemen in de zandgebieden van Nederland* [Landscape planning and water systems on sandy soils in the Netherlands]. PhD thesis, Wageningen University, Wageningen.

van der Schaaf, S. and J.P.M. Witte, 1997*, Inleiding Landschapshydrologie* [Introduction Landscape hydrology]. Wageningen University, Department of Water Resources, Reader, Wageningen.

van der Sluijs, P., 1982, De grondwatertrap als karakteristiek van het grondwaterstandsversloop. *H_2O*, 15(3):42-46.

van Diggelen, R., 1998, *Moving gradients: assessing restoration prospects of degraded brook valleys*. PhD thesis, Wageningen University, Wageningen.

van Drecht, G., 1997, Modellen voor diffuse ontwatering in de toplaag [Models of diffuse drainage in the upper layer]. *Stromingen* 3(2):5-16.

van Ek, R., J.P.M. Witte, J. Runhaar and F. Klijn, 2000, Ecological effects of water management in the Netherlands: the model DEMNAT. *Ecological Engineering* 16:127-141.

van Ierland, E.C. and N.Y.H. de Man, 1993, Ecological engineering: first steps towards economic analysis. *Ecological Engineering*, 7:351-371.

van Staalduinen, L.C., M.W. Hoogeveen, C. Ploeger and J. Dijk, 1996, *Heffing van grondwaterbelasting via een forfait: een onderzoek naar de mogelijkheden voor de land- en tuinbouw* [Levying a groundwater tax]. Agricultural Economics Research Institute, Publication 3.163, The Hague.

Wichelns, D., 1999, An economic model of waterlogging and salinisation in arid regions. *Ecological Economics*, 30(3):475-491.

Wiersma, D., 1998, Towards a more sustainable use of groundwater by means of economic incentives. A case study of The Netherlands. In: S. Dwyer, U. Ganslosser and M. O'Connor (eds.), *Life Science Dimensions: Ecological Economics and Sustainable Use*. Filander Verlag, Fürth.

Winpenny, J., 1994, *Managing Water as an Economic Resource.* Routledge, London and New York.

Witte, J.P.M., 1998, *National water management and the value of nature*. PhD thesis, Wageningen University, Wageningen.

Witte, J.P.M. and F. Klijn, 1997, Waardering van standplaatstypen: vuistregels voor een beoordeling van potentiële botanische natuurwaarden [Valuation of site types: rules of thumb to value potential botanical nature values]. *Landschap*, 14(2):105-109.

Witte, J.P.M., J. Runhaar and R. van Ek, 2001, Three approaches to ecohydrological modelling. In: International Hydrological Programme, *Ecohydrology – from concepts to applications*. Cambridge University Press, Cambridge.

Working group HELP [Werkgroep HELP], 1987, *De invloed van de waterhuishouding op de landbouwkundige productie* [The impact of water management on agricultural production]. Mededelingen Landinrichtingsdienst 176, Utrecht.

World Bank, 1993, *Water resources management: a World Bank policy paper*. World Bank, Washington.

Young, R.A. and R.H. Haveman, 1985, Economics of water resources: a survey. In: A.V. Kneese et al. (eds.), *Handbook of natural resource and energy economics*. North-Holland P.C., Amsterdam.

Zhang, Y., 2001, Economics of transaction costs saving forestry. *Ecological Economics*, 36:197-204.

Zilberman, D., U. Chakravorty and F. Shah, 1997, Efficient management of water in agriculture. In: D.D. Parker and Y. Tsur (eds.), *Decentralization and co-ordination of water resource management.* Kluwer Academic Publishers, Boston.

Zilberman, D. and L. Lipper, 1999, The economics of water use. In: J.C.J.M. van den Bergh (ed.), *Handbook of environmental and resource economics*. Edward Elgar, Cheltenham.

Zilberman, D., N. MacDougall and F. Shah, 1994, Changes in water allocation mechanisms for California agriculture. *Contemporary Economic Policy*, XII:122-133.

Zilberman, D., M. Wetzstein and M. Marra, 1993, The economics of non-renewable and renewable resources. In: G.A. Carlson, D. Zilberman and J.A. Miranowski (eds.), *Agricultural and environmental resource economics*. Oxford University Press, New York.

List of main symbols used

Symbol	Definition	Dimension	Chapter
A	agricultural surface area	L^2	3
A^a	desiccated (affected) surface area nature reserve	L^2	4
A^n	agricultural surface area with special ecological value	L^2	3
A^t	surface area nature reserve	L^2	4
A^u	unaffected surface area nature reserve	L^2	4
B	weight attached to restoration of the nature reserve	-	4
C	costs to agriculture (of nature restoration)	€T^{-1}	4
C^d	agricultural drought damage costs	€T^{-1}	4
C^w	agricultural wet damage costs	€T^{-1}	4
c	hydraulic resistance of the top layer	T	4
c_1	vertical resistance aquitard	T	4
c^{dran}	drainage resistance farmland	T	4
c^{vert}	vertical resistance farmland	T	4
D	groundwater depth relative to soil surface	L	2,3,4
D^h	highest groundwater depth relative to soil surface	L	4
D^l	lowest groundwater depth relative to soil surface	L	4
d	drought damage fractions of potential crop yield	-	4
E, $\boldsymbol{E}$	agricultural shallow groundwater extraction, i.e. irrigation rate	LT^{-1}, $\mathbf{L^3T^{-1}}$	1,2,**5**
E^A	cost-effectiveness of an increase in the unaffected surface area	L^2€$^{-1}$	4
E^V	cost-effectiveness of an increase in the conservation value	cvu€$^{-1}$	4
$\boldsymbol{e}$	effective water	L^3T^{-1}	5
F	fractions of farmland with crop I	-	4
G, $\boldsymbol{G}$	capillary rise rate	LT^{-1}, $\mathbf{L^3T^{-1}}$	1,**5**
g	efficiency gains	€T^{-1}	6
GWT	Ground Water Table class	-	4
H^f	groundwater level (height) relative to the surface water level	L	4,5
H^l	hydraulic groundwater head relative to the surface water level	L	4
h	irrigation efficiency (fraction of applied water utilised by the crop)	-	5
I	precipitation rate	LT^{-1}	1
J	evapotranspiration rate	LT^{-1}	1
K	technology cost	€L^{-2}	5
k_{hor}	horizontal hydraulic conductivity phreatic aquifer	LT^{-1}	4
k_{vert}	vertical hydraulic conductivity phreatic aquifer	LT^{-1}	4
L	length of the nature reserve	L	4
l	level of damage to environmental amenities	-	5
M	distance between the top of the aquitard and the ditch water level	L	4
MC	marginal costs	€ T^{-1}	2,6
MT^c	marginal transaction costs of policy instruments	€ T^{-1}	6
N	nature index	-	3

N^{R}	nitrate concentration in recharge flows	ML^{-3}	5
N^{S}	nitrate concentration in the groundwater stock	ML^{-3}	5
O, $\boldsymbol{O}$	percolation rate	LT^{-1}, $\mathbf{L^3T^{-1}}$	1,**5**
P	price of the crop	€M^{-1}	3,4,5
p	occurrence probabilities	-	4
Q	reduction in agricultural groundwater extraction	$L^3\,T^{-1}$	6
R, $\boldsymbol{R}$	groundwater recharge rate	LT^{-1}, $\mathbf{L^3\,T^{-1}}$	2,4,**5**
$\boldsymbol{S}$	groundwater quantity (stock size)	$\mathbf{L^3}$	**5**
S^{s}	soil surface level	L	4
ΔS_{sa}	change in water storage of the saturated zone	LT^{-1}	1
ΔS_{to}	change in water storage of the combined unsaturated-saturated zone	LT^{-1}	1
ΔS_{un}	change in water storage of the unsaturated zone	LT^{-1}	1
s	slope of the nature index	-	3
T	transmissivity of the underlying aquifer	L^2T^{-1}	4
T^{c}	transaction costs of policy instruments	€$L^{-3}\,T^{-1}$	6
T^{v}	volumetric tax	€L^{-3}	2,5
U	regional upward seepage rate	LT^{-1}	4
U_{in}	upward seepage rate	LT^{-1}	1
U_{out}	downward seepage rate	LT^{-1}	1
V	annual monetary value of nature	€$L^{-2}\,T^{-1}$	3,5
V^{g}	annual conservation value of each Ground Water Table class	cvu T^{-1}	4
V^{r}	annual conservation value of the nature reserve	cvu T^{-1}	4
V^{s}	annual conservation value of the site types	cvu T^{-1}	4
V_{in}	inlet rate foreign water	LT^{-1}	1
V_{out}	drainage rate	LT^{-1}	1
v	price of agricultural groundwater extraction/irrigation water	€L^{-3}	2,5,6
W	width of the nature reserve	L	4
w	wet damage fractions of potential crop yield	-	4
X	distance to the middle of the nature reserve	L	4
Y	yield of the crop	ML^{-2}	3,4,5
Z	compensation payments	€ T^{-1}	4
ϕ	ratio between the affected nature reserve and farmland irrigated	-	5
α, β, δ	parameters	-	3,4,6
λ	leakage factor	L	4
μ	current value shadow price of changes in stock quality over time	€	5
ν	current value shadow price of changes in stock quantity over time	€	5
π	quasi-rent	€L^{-2}	5
ρ	discount rate	-	5
σ	standard deviation of the soil surface level	L	4
υ	average soil surface level	L	4

Curriculum Vitae

Petra Jacoba Gerarda Johanna Hellegers werd op 11 februari 1971 te Boxmeer geboren. In 1989 behaalde zij haar VWO-diploma aan het Elzendaalcollege te Boxmeer. Aansluitend studeerde zij Agrarische Economie aan de toenmalige Landbouwuniversiteit te Wageningen. In augustus 1993 studeerde zij af. Van augustus 1993 tot februari 1994 was zij werkzaam bij de toenmalige vakgroepen Algemene Agrarische Economie en Ontwikkelingseconomie.

Sinds februari 1994 werkt zij als wetenschappelijk onderzoeker bij het LEI te Den Haag. In 1998 begon zij daar met haar promotieonderzoek, dat ondergebracht werd bij de Mansholt Graduate School of Social Sciences. Voor dit onderzoek verbleef zij in 1999 gedurende vier maanden als Associate Researcher aan de University of California te Berkeley. In 1999 behaalde zij het diploma van het Netwerk Algemene en Kwantitatieve Economie (NAKE).

Resultaten van dit onderzoek zijn/worden gepubliceerd in de wetenschappelijke tijdschriften ‘Journal of Environmental Planning and Management’ (Hoofdstuk 3), ‘Ecohydrology and Hydrobiology’ (Hoofdstuk 4), ‘Ecological Economics’ (Hoofdstuk 5) en ‘Water Resources Update’ (deel van Hoofdstuk 6) en in het boek ‘Economics of Water Resources: Theory and Policy’ (deel van Hoofdstuk 6). Ook werden resultaten gepresenteerd op internationale congressen, o.a. van de EAERE in Oslo, de EAAE in Warschau, de AAEA in Chicago en op waterconferenties in Cyprus en Spanje en de World Water Fair in Den Haag.

Na vier jaar studie, vier jaar toegepast onderzoek en vier jaar promotieonderzoek zet zij de komende jaren haar activiteiten voort o.a. als projectleider van het project ‘Water valuation and pricing’, dat onderdeel is van het programma ‘Partners in Water for Food’. Hierbij wordt samengewerkt met het IFPRI in Washington en IWMI in Colombo.

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