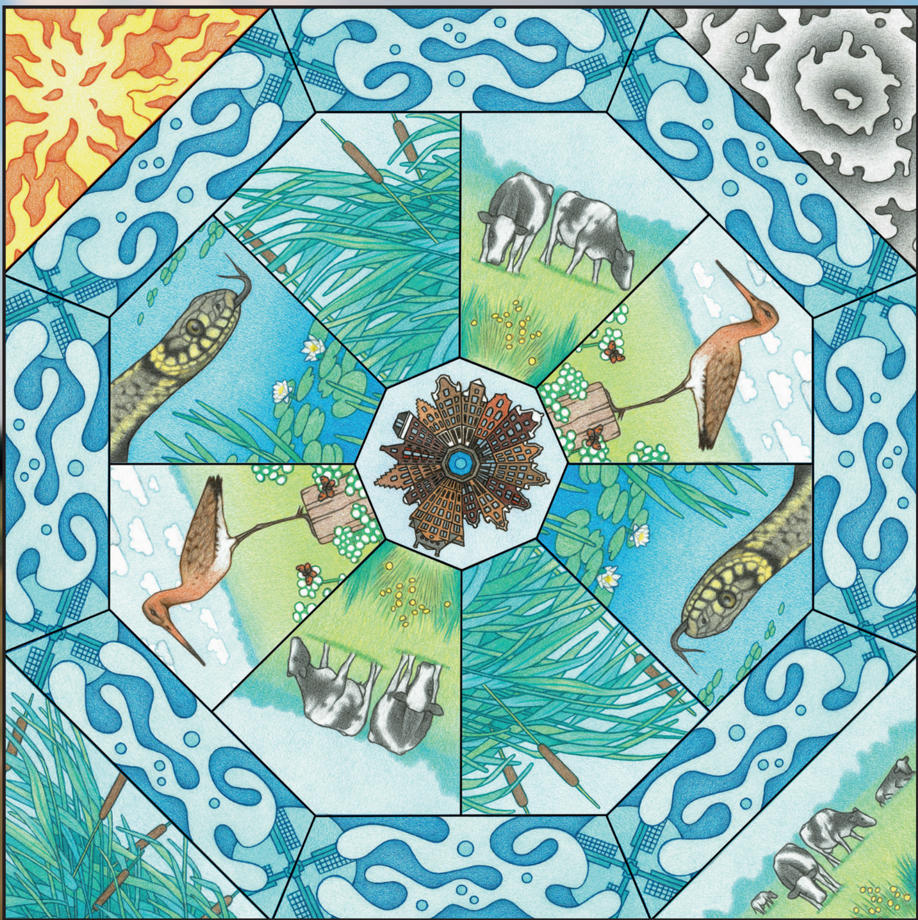
Informed science–policy interactions

Advancing the support of collaborative management of social–ecological systems



Henk van Hardeveld

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Informed science–policy interactions

Advancing the support of collaborative management of social–ecological systems

Geïnformeerde kennis–beleid interacties

Naar een verbetering van de ondersteuning van gezamenlijke beleidsontwikkeling voor sociaal–ecologische systemen

(met een samenvatting in het Nederlands)

Proefschrift

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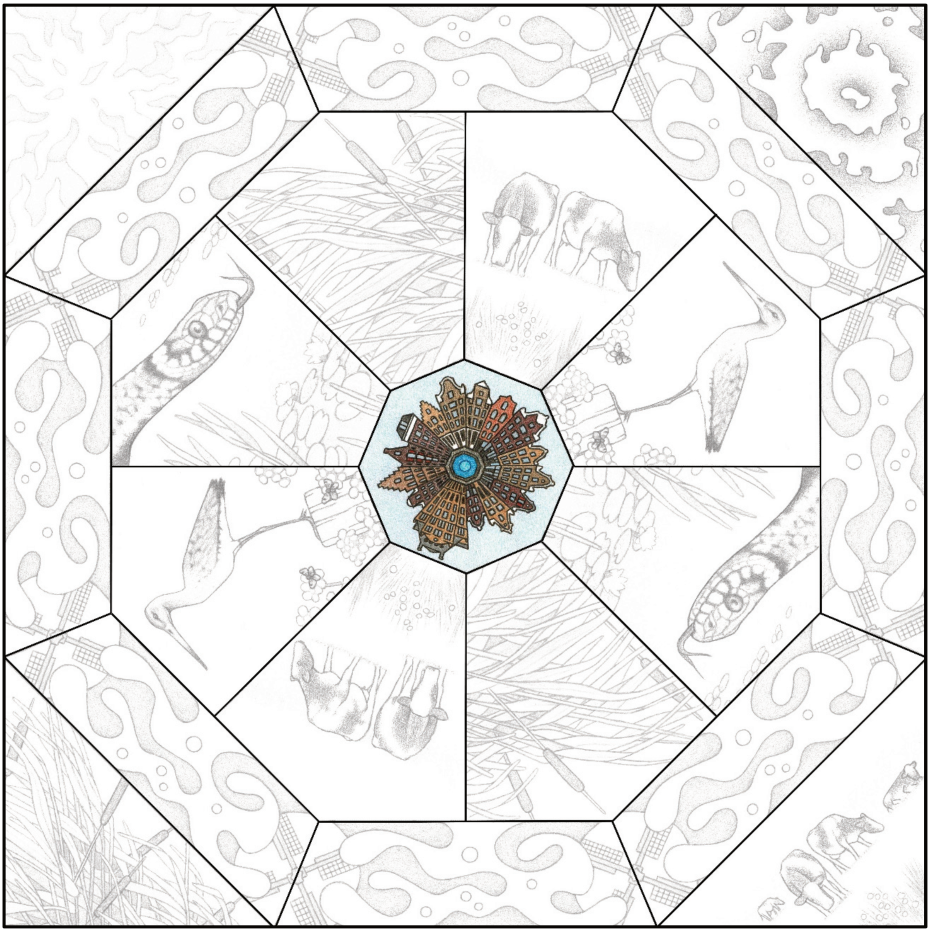

Ecological, economic and social systems are closely intertwined, resulting in complex, social–ecological systems. To manage them sustainable, we must not only understand the biophysical dynamics of the ecosystem, but also the social dynamics of the users, and the organizations and rules that are created to govern the system's use. The users and their organizations and rules are depicted by the houses in the center of the figure.

1.1 Background

How to embrace the complexity of social–ecological systems?

On the brink of the third decade of the twenty-first century, humanity is still struggling with the rampant deterioration of its environment. In fact, the current magnitude of pollution, emission of greenhouse gasses, land degradation, over-exploitation of natural resources, and loss of biodiversity is unprecedented in the span of human history (UN Environment, 2019). Moreover, if the unsustainable production and consumption patterns continue unchecked, the ecological foundations of our society may be depleted beyond recovery. Therefore, one of the most poignant questions of our time is how to arrest and reverse the environmental degradation, and how to realize a more sustainable use of our remaining natural treasures.

There is no easy answer to this question. Ecological, economic and social systems are closely intertwined, resulting in complex, social–ecological systems (SESs). SESs are composed of a biophysical resource system and the units and services it provides, the users of the system, and the organizations and rules that are created to govern the system's use (Ostrom, 2009). All these subsystems interact to produce outcomes across multiple spatial and temporal scales that are often nonlinear and highly uncertain. It is therefore widely acknowledged that to achieve a more sustainable management of SESs, we must move beyond panaceas and instead adopt a perspective that embraces complexity (Ostrom, 2007). We must do so in a way that is as simple as possible but no simpler than is required for our understanding and communication; this should be done in a dynamic and descriptive way that connects our recognition of the past and present to policies and evaluations of different future scenarios (Holling, 2001).

In response to SES complexity, management approaches are increasingly focused on resilience, i.e., a system's ability to maintain its functionality or to regenerate after disruptive changes (Walker et al., 2002). To manage the resilience of SESs, adaptive management approaches have emerged that propagate a structured and iterative learning-by-doing strategy (Folke, 2006; Den Uyl and Driessen, 2015). Initially, these approaches tended to focus on enhancing the scientific knowledge of the SES. However, because the knowledge generated was frequently not successfully linked to management, more iterative approaches were designed that allowed stakeholders to collaborate (Walker et al., 2002; Folke, 2006; Scarlett, 2013). The collaboration of stakeholders may have multiple benefits, derived from (a) normative ideas and principles such as the enhancement of democratic capacity or deliberation among participants, (b) a substantive rationale to improve the quality of decisions, and (c) an instrumental underpinning to generate legitimacy or resolve conflict (Glucker et al., 2013). It is suggested that the appropriate level of stakeholder collaboration increases as the complexity of SES management increases, and when transformative rather than incremental changes are needed (Beratan, 2014).

From this collective body of scientific literature, the suggestion emerges that collaborative management is at least part of the answer to the question how we can use SESs more sustainably. However, collaborative management has proven difficult to put into practice, due to stakeholders who are unwilling to cooperate, a lack of capacity for environmental management among stakeholders, and knowledge that is context-dependent. To alleviate this predicament, social learning processes have been advocated, aimed at "learning together to manage together" (Folke, 2006; Pahl-Wostl et al., 2007; Monroe et al., 2013).

Social learning takes place through social interactions and processes between actors within networks, resulting in a change in understanding at the individual level as well as the group level, for example within Communities of Practice (Reed et al., 2010).

<u>Managing the interface between science and policy</u>

Although social learning is essentially defined as a social process, we must take care not to over-emphasize the social features, but duly consider the scientific features too. Van de Riet (2003) points out that if a process focuses too much on social features such as gaining the support of the participating stakeholders, this may result in "negotiated nonsense", i.e., knowledge that is supported by stakeholders but is scientifically invalid. Simultaneously, she points out that an over-emphasis of the scientific features may result in "superfluous knowledge", i.e., knowledge that is scientifically valid, but irrelevant to the management problem. The latter problem has occurred quite often, with scientists focusing on increasing the supply of credible scientific information without considering its relevance and usefulness for decision-makers. To produce useful knowledge for collaborative management, i.e., knowledge that is scientifically valid and relevant to the management practice, social learning processes must "navigate between negotiated nonsense and superfluous knowledge" (Van de Riet, 2003).

To effectively produce useful knowledge, interfaces are needed that simultaneously enhance the salience, credibility, and legitimacy of the scientific information (Cash et al., 2003). Therefore, sufficient attention must be given to the process of identifying which information is most useful for decision-making, thus reconciling the supply of scientific information with users' demands (McNie, 2007). Various organizational arrangements have been suggested to manage the interface between science and practice, encompassing individual "boundary-spanning mediators", processes of participatory knowledge development, and boundary organizations (Cash et al., 2003; McNie, 2007; Van Enst et al., 2014). Collectively, these arrangements have come to be known as "science–policy interfaces" (SPIs), i.e., organizational arrangements for processes of social interaction between scientists and other actors in the policy process, which enhance the exchange, co-evolution, and joint construction of knowledge, with the aim of enriching decision-making (Van den Hove, 2007; Van Enst, 2018).

All types of SPIs that are harnessed to enrich the collaborative management of SESs have one thing in common: they aim for the production of useful knowledge, i.e., a carefully balanced intermediate state between superfluous knowledge and negotiated nonsense, where science effectively meets the information demands of practitioners, who then interactively make scientifically underpinned management decisions. In this thesis, this shared goal of SPIs is called "informed science–policy interaction". Unfortunately, achieving informed science–policy interaction is not easy because many SPIs encounter interaction problems that diminish their effectiveness. Van Enst et al. (2014) point out how strategic production and/or use of knowledge will negatively affect the credibility and legitimacy of knowledge. This negative consequence mainly occurs when the knowledge is uncertain and/or consensus on norms and values is lacking. As a result, so-called "dialogues of the deaf" may evolve, i.e., deadlocks in which each participant advances their own arguments without listening to those of others (Van Eeten, 1999).

Another type of interaction problem that SPIs may encounter is the operational misfit between the demand and supply of knowledge, which will reduce the salience of information (Van Enst et al., 2014). To facilitate communication, translation, and mediation across boundaries, many SPIS use "boundary objects", i.e., collaborative outputs that "are both adaptable to different viewpoints and robust enough to maintain identity across them'' (Star and Griesemer, 1989); these include decision support systems and simulation models (White et al., 2010), GIS technology (Harvey and Chrisman, 1998), and multi-faceted concepts such as "ecosystem services" (Abson et al., 2014). Uran and Janssen (2003) describe how many decision support systems were not effective boundary objects because they failed to provide their users with salient information. In addition, Van Kouwen et al. (2007) evaluate the applicability of decision support systems, and they identify a range of challenges regarding (a) knowledge, i.e., the integration of socio-economic and biophysical system dynamics, as well as the simulation of spatial and temporal system dynamics, and (b) processes, i.e., the involvement of stakeholders, and the ability to make complex information understandable and interactively accessible. Remarkably, they point out that none of the decision support systems they evaluated were able to meet all challenges.

These findings illustrate that the effectiveness of an SPI is context-dependent. The characteristics of the SES and the related management challenges determine what type of tools and processes are useful. Goosen et al. (2007) describe a continuum of decision support tools, ranging from a focus on interaction to a focus on analysis. Tools focusing on interaction aim at revealing preferences, specifying objectives and defining alternatives; examples include games and exercises. Tools focusing on analysis are data-driven and aim at evaluating performance or ranking alternatives; examples include Cost–Benefit Analysis (CBA) and scenario analyses. As the usefulness of decision support tools is context-dependent, it has been suggested that we should not pursue one-size-fits-all solutions but rather focus our efforts on increasing our collective understanding of which approaches work in various situations (McNie, 2007; Van Enst et al., 2014). Furthermore, special emphasis should be placed on increasing the usefulness of the knowledge exchanged between scientists and practitioners (Beratan, 2014). Apparently, although SPIs have been amply discussed in the scientific literature, a clear understanding of how to design and implement them is a persistent gap in our knowledge.

<u>Aim and research question</u>

To fill this knowledge gap, this thesis aims to contribute to our collective understanding of how to design and implement SPIs for the collaborative management of SESs. This understanding can help us overcome implementation challenges and guide us towards the informed science–policy interactions which are needed to achieve a more sustainable use of our remaining natural resources. The collaborative management process regarding the future of the Dutch peatlands serves as an illustrative context for this endeavor. The main research question is: *"How can science–policy interfaces be designed and implemented to advance the collaborative management of social–ecological systems?"*

1.2 The Dutch peatlands

<u>Peatland dynamics throughout the centuries</u>

As the name indicates, the Netherlands are located in the nether parts of the dynamic Rhine-Meuse delta. Consequently, the landscape has changed substantially in the Holocene (Vos et al., 2011). When the groundwater table gradually rose with post-glacial sea level rise, thick eutrophic wood-sedge peat deposits accumulated, covering vast parts of the delta around 500 BC (Fig. 1.1A). However, in the subsequent millennia, many of these deposits were eroded (Fig. 1.1B). First, the sea reached further inland from the north, reaching the freshwater lakes in the central part of the Netherlands, and gradually eroding the surrounding peat deposits. In addition, due to deforestation in the Roman era, the sediment load of the rivers increased, resulting in clay deposits on top of the peat deposits adjacent to the rivers. Furthermore, storm surges caused several devastating floods, eroding vast areas of peat deposits. As a consequence of these processes in this deltaic environment, the area of peat deposits was significantly reduced during Roman and medieval times.

After approximately 1000 AD, humans increasingly influenced the landscape and the water system. Due to peat exploitation and excavation, large freshwater lakes emerged in the western and northern parts of the peatlands. As a result of erosion by waves, these lakes gradually became larger (Fig. 1.1B). In addition, in the thirteenth century the first water authorities were founded, enabling active management of the natural watercourses. The water authorities leveed vast areas along the coast and the rivers. Furthermore, they created artificial catchments – polders – with a dense network of ditches and traditional windmills, thus allowing the natural fens to be converted to agricultural fields and meadows. It is noteworthy that these medieval water systems and allotment patterns still exist and are acclaimed as valuable Dutch cultural heritage.

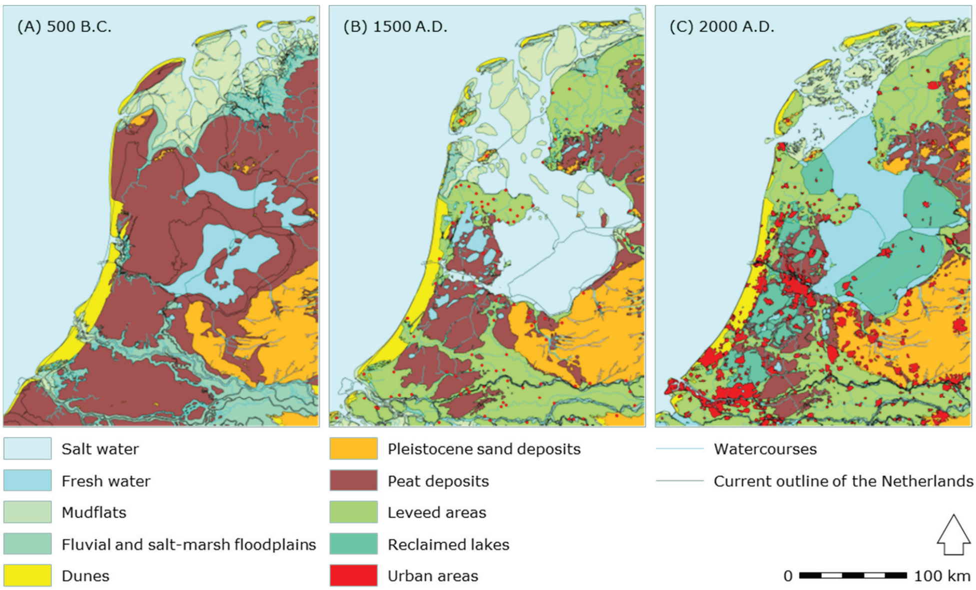

Fig. 1.1. Paleogeographic maps of the Rhine-Meuse delta, reflecting a decline in peat deposits since 500 BC due to natural and anthropogenic processes (after Vos and De Vries, 2013).

The agricultural use of the peatlands required drainage to lower the groundwater table and allow oxygen to enter the plant root zone. At first, excavating the ditches was sufficient to drain the peat soils. However, lowering the groundwater table caused consolidation and shrinkage of the peat soil, and the increased oxygen availability in the soil led to peat oxidation. Consequently, drainage caused the peat soils to subside. At the end of the Middle Ages, due to continuing soil subsidence, the groundwater tables had become higher relative to the ground surface. Some scholars believe that these wetter circumstances caused the dominant land-use in the Dutch peatlands to change from the cultivation of cereal crops to dairy farming, whereas others suggest this change was more likely instigated by changed economic circumstances (Tielhof and van Dam, 2006). These scholars point out that at this time, it was already possible to lower the surface water levels further by using windmills. However, the population in Europe had markedly decreased by then, and as a consequence, the market for cereal crops had collapsed, and investment in windmills would not have been profitable.

In the subsequent centuries, windmills became a frequently occurring feature of the Dutch peatlands after all. From the sixteenth century onwards, windmills enabled the reclamation of the freshwater lakes in the western part of the Netherlands. In addition, they were used to lower the drainage levels in peatlands, enabling the excavation of peat. Although people were aware that the excavation of peat had devastating effects to the landscape, short-term economic gains usually prevailed over these long-term environmental concerns (Tielhof and Van Dam, 2006). As a result, large parts of the remaining peatlands were irreversibly lost. In addition, the progressive lowering of the drainage levels resulted in a cumulative soil subsidence over 8–10 centuries of 2–4 m (Schothorst, 1977; Van Asselen et al., 2018). This resulted in the loss of thin peat deposits, especially in the central and northern parts of the Netherlands. Furthermore, during the past century, the remaining area of peatland has been further diminished by continuing urbanization. As a consequence of all these anthropogenic processes, the current area of peatland is significantly smaller than it was several centuries ago (Fig. 1.1C). This sharp decline in peatland area is not limited to the Netherlands but is also present in most European countries with peat deposits (Bragg and Lindsay, 2003).

Governing the peatlands

The overview of the history of the Dutch peatlands clearly illustrates the biophysical and socio-economic dynamics of the SES. The modes of environmental management that evolved during the preceding centuries further contribute to the singularity of the peatlands. The management of peatlands has always been a joint effort of stakeholders and governmental organizations, because the livelihood of most people directly depended on it. Most of the time, water management was geographically limited to a polder area. In response to this, a consensus-based socioeconomic model for policy making evolved, which in the late twentieth century became globally known as the Dutch "Polder model". A key characteristic of this model is the recognition that despite their differences, all the involved stakeholders need each other and therefore some form of consensus must be reached. To understand the background of the Polder model that evolved in the Dutch peatlands, it is important to consider not only the process of stakeholder participation but also the entire environmental governance context. The participation of stakeholders in the management of an SES reflects "a process whereby individuals, groups, and/or organizations choose to take an active role in decision-making processes that affect them" (Reed et al., 2010). Environmental governance encompasses a collection of features regarding (a) actor configurations, such as stakeholder positions and power bases, (b) the institutions that shape the interactions between the actors, and (c) the content of these arrangements, in terms of goals, preferred policy instruments, and characteristics of the SPI employed (Driessen et al., 2012).

The water authorities that were founded in the Middle Ages still exist, which makes them one of the oldest operational branches of government in the entire world. In recent decades, they have changed from primarily local organizations to professional regional institutions that are able to contribute to broader environmental management tasks. To cope with these tasks, many local water authorities have merged, resulting in a decrease from several thousand to 21 regional organizations. In addition, water authorities increasingly employ scientifically educated advisors, and they collaborate more closely with organizations specialized in the application of scientific knowledge (Blankesteijn, 2011). Van der Brugge et al. (2005) suggest that this development reflects a transition from a technocratic-scientific to an integrated-participatory water management style. However, Driesssen et al. (2012) point out that modes of environmental governance tend to build on each other rather than replace one another. They propose that environmental governance is often multi-faceted, with simultaneously co-existing forms. When their framework for comparing modes of environmental governance is applied to the current situation in Dutch peatlands, a multi-faceted picture emerges. On the one hand, there are features of a centralized mode of governance, such as the primacy given to agricultural land use in the peatlands, exemplified by the subsidies that are granted because of the Common Agricultural Policy of the European Union. On the other hand, there are also features of an interactive mode of governance, i.e., a mode whereby governments and stakeholders from civil society parties both play an active role and jointly arrive at a decision (Driessen et al., 2001; Edelenbos, 2005). For example, processes of adaptive management and transition management which result in negotiated project agreements (Den Uyl and Driessen, 2015).

At a national level, the stage for the debate regarding the future of the Dutch peatlands was set in the wake of World War II. At that time, the Common Agricultural Policy of the European Union was set up, to ensure sufficient food at all times, preventing famines such as had occurred during World War II. In the Dutch peatlands, re-allotment processes established the conditions for more productive agriculture. These conditions included lower water levels, which increased the soil subsidence rates. At the end of the twentieth century, a slightly more integrated policy was implemented, aimed at slightly higher water levels, and compromising between maintaining good conditions for agricultural production and slowing down the soil subsidence rates. In addition, several projects were instigated that were aimed at a top-down implementation of a rise in water levels, which was accompanied by a transition from dairy farming to nature restoration. These projects were fueled by a CBA which claimed that the loss in Net Value Added by dairy farming would be outweighed by decreasing management costs for the water system and the infrastructure of roads and sewers (Van Brouwers-Haven and Lokker, 2010). However, the projects met with fierce resistance from agricultural stakeholders. The CBA was contested because it ignored not only distributional effects but also the farmers' adaptive capacity to increase the Net Value Added of agricultural production in the peatlands. This lock-in situation made governmental organizations aware that more effective stakeholder collaboration was needed to develop viable, collaborative management options.

In the first two decades of the twenty-first century, several SPIs were instigated in the Dutch peatlands. At the national and regional level, scientists and societal stakeholders united in various processes of participatory knowledge development regarding the future of the Dutch peatlands (Van Brouwershaven and Lokker, 2010) and strategies for climate adaptation in the peatlands (Brouns et al., 2015; Driessen et al., 2015). In addition, a boundary organization was created for innovative peatland management. These SPIs have produced generic knowledge, raised awareness of adaptations, and provided guidance for regional adaptation strategies. However, at the local level, some uncertainty still remains regarding site-specific effects from particular stakeholder perspectives. It is essentially this kind of knowledge that SPIs for interactive governance should focus on (Driessen et al., 2012). This observation suggests there is a need to design and implement SPIs that can support the collaborative management of the Dutch peatlands at a local level, where science meets not only general policies, but also a multi-facetted reality, in which stakeholders are faced with the challenge to put these policies into practice.

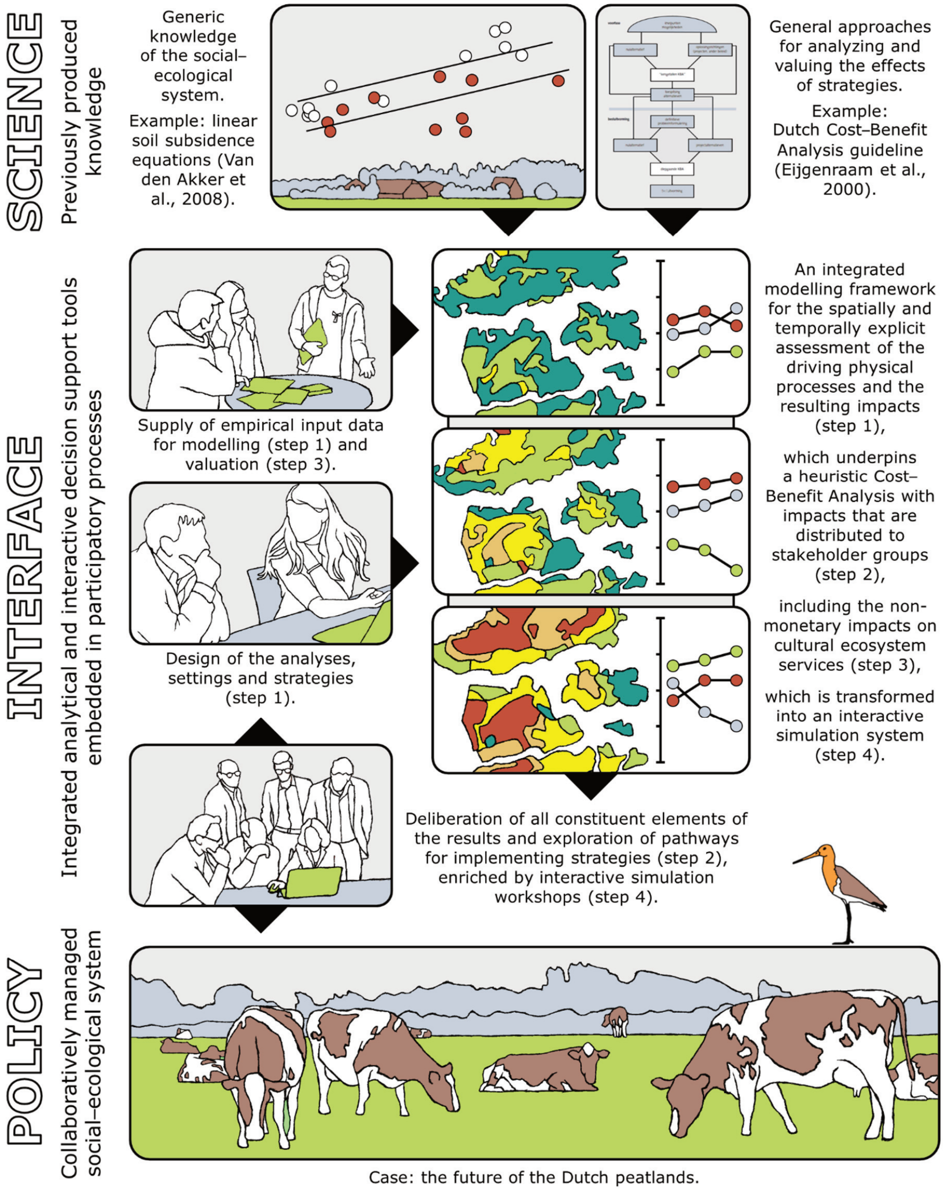

Fig. 1.2. Overview of the science–policy interface that is developed in this thesis, consisting of integrated analytical and interactive decision support tools embedded in participatory processes. It is developed in four sequential steps and implemented in a collaboratively managed social–ecological system, i.e., the Dutch peatlands, for which it translates previously produced knowledge into site-specific effects from multiple stakeholder perspectives. Chapters 2–4 present the background of the previously produced science. Section 1.2 addresses the Dutch peatlands. The interface is discussed and evaluated throughout all chapters. The triangles indicate the direction in which the features of the interface feed into each other. The steps in which the features are introduced are given in parentheses.

Meanwhile, the stage that was set in the wake of World War II has remained essentially unchanged; in other words, the priority given to agricultural production still prevails. However, this status quo has increasingly been challenged in recent years, fueled by growing concerns over the emission of greenhouse gasses, the loss of biodiversity, and the impact of soil subsidence in general. This raises the question whether this indicates that, on the brink of the third decade of the twenty-first century, new opportunities are emerging to arrest and reverse the environmental degradation of the Dutch peatlands, thus paving the way for the collective stakeholders to use their valued SES more sustainably. If so, SPIs that can support this endeavor are now more necessary than ever before.

1.3 Approach

General approach

The general approach is to integrate analytical and interactive decision support tools, embed them in participatory processes, implement the resulting SPI in a real-world case study regarding the future of the Dutch peatlands, and evaluate the SPI's contribution to the support of informed science–policy interactions. This is illustrated in figure 1.2. The development of the SPI consists of four sequential steps (Table 1.1). The step-by-step development allows for a comprehensive evaluation of the usefulness of its constituent features. Moreover, the step-by-step feedback from real-world stakeholders helps to identify features that can further improve the usefulness of the SPI, providing guidance to overcome implementation challenges. The first step (Chapter 2) focuses on the scientific backbone of the SPI. It creates an integrated modelling framework which transforms the generic knowledge of the biophysical dynamics of the Dutch peatlands that was produced in previous SPIs into spatially and temporally explicit, site-specific knowledge. Steps 2 and 3 (Chapters 3 and 4) combine the SPI of step 1 with general frameworks for analyzing the SES dynamics and valuing the effects of management strategies, in particular a heuristic CBA (step 2) and a valuation of cultural ecosystem services (step 3). All steps consider the stakeholders of the peatlands as well as their interactions with each other and the peatlands in general. Step 4 (Chapter 5) places special emphasis on these interactions, integrating all previous features into an interactive simulation system (ISS).

Table 1.1. The sequence in which decision support tools and participatory processes are added to the science–policy interface that is developed in this thesis. Note that tools and processes are preserved in the subsequent steps of the development of the science–policy interface.

Step	Decision support tools	Participatory processes	Chapter
1	• An integrated modelling framework for the spatially and temporally explicit simulation of the driving physical processes and their resulting impacts	• Supply of empirical input data for the integrated modelling framework • Co-design of the analyses • Design of settings and strategies	2
2	• A heuristic Cost–Benefit Analysis with impacts that are distributed to stakeholder groups	• Deliberation of all constituent elements of the results and exploration of pathways for implementing strategies	3
3	• Assessment of non-monetary impacts on ecosystem services	• Supply of empirical input data for the valuation of cultural ecosystem services	4
4	• Interactive simulation system	• Interactive simulation workshops	5

Step 1: An integrated modelling framework

The first step in the development of the SPI (Chapter 2) focuses on the assessment of the overriding problem of the Dutch peatlands, namely soil subsidence. The aim is to develop the analytical core of the SPI, which can then serve as the foundation for subsequent versions of the SPI. Therefore, this step reflects the development of an integrated modelling framework that simulates the interrelated dynamics of water management and soil subsidence and that determines the spatial and temporal range of societal impacts. The stakeholders of the Dutch peatlands participate in the development of the modelling framework by supplying empirical input data, co-designing the societal impact assessments, and defining the settings and strategies that are assessed in the modelling framework (Fig. 1.3). The guiding research question is: *"Which long-term impact models are useful for supporting management strategies in peatlands, and how they can be integrated?"*

The development of the modelling framework starts by using previously produced generic scientific knowledge to underpin a GIS-based model that simulates the governing physical processes of the peatlands, i.e., water level dynamics and soil subsidence. Subsequently, stakeholders and scientists co-design several additional GIS models that allow for an integrated assessment of all the relevant impacts of management strategies. The design of the modelling frameworks enables spatially and temporally explicit impact assessments, which can be distributed to all the stakeholder groups involved. The usefulness of the modelling framework is demonstrated by assessing the impact of long-term management strategies in part of the Dutch peatlands and by evaluating the relevance of the results for improving the collaborative management of the peatlands.

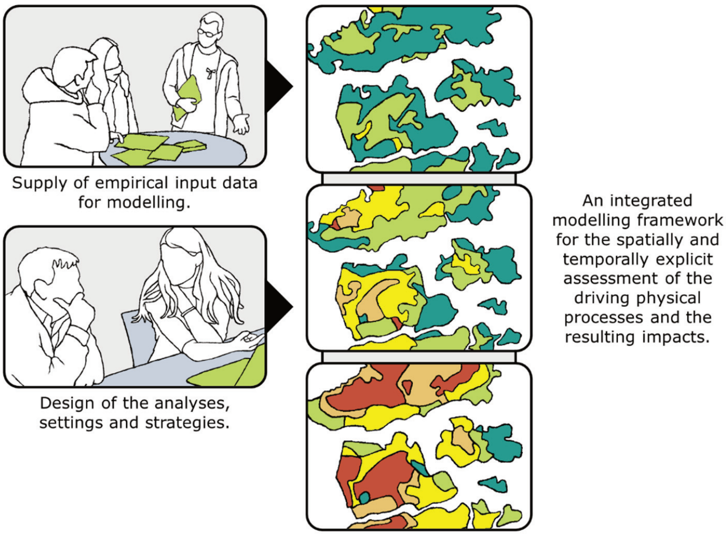

Fig. 1.3. General design of the science–policy interface of step 1. Triangles indicate the direction in which the features feed into each other. See Fig. 2.2 for a more detailed design of the modelling framework.

Step 2: Cost–Benefit Analysis as a heuristic aid

The second step in the development of the SPI (Chapter 3) focuses on organizing and discussing the information that is provided by the integrated modelling framework. It reflects a combination of the integrated modelling framework with a CBA with enhanced discursive features. The CBA is not used as a decision-rule to determine the optimal cost–benefit ratio of project alternatives, but as a heuristic aid, aiming for an intersubjective assessment of the project alternatives. To facilitate this, final verdicts regarding the optimal cost–benefit ratio of management alternatives are avoided. Instead, costs and benefits are distributed to the stakeholder groups involved, and all the constituent elements of the results are presented. Although many scholars believe such an approach has high potential to support collaborative policy processes (De Jong and Geerlings, 2003; Turner, 2007; Browne and Ryan, 2011; Robinson and Hammitt, 2011; Beria et al., 2012), case studies that demonstrate the added value remain underexposed in the scientific literature, especially because many CBAs encounter process issues that inhibit their effectiveness as a decision support tool (Beukers et al., 2012; Mouter et al., 2013). Step 2 fills this knowledge gap. The guiding research question is: *"What is the added value of a heuristic CBA for the support of the collaborative management of SESs?"*

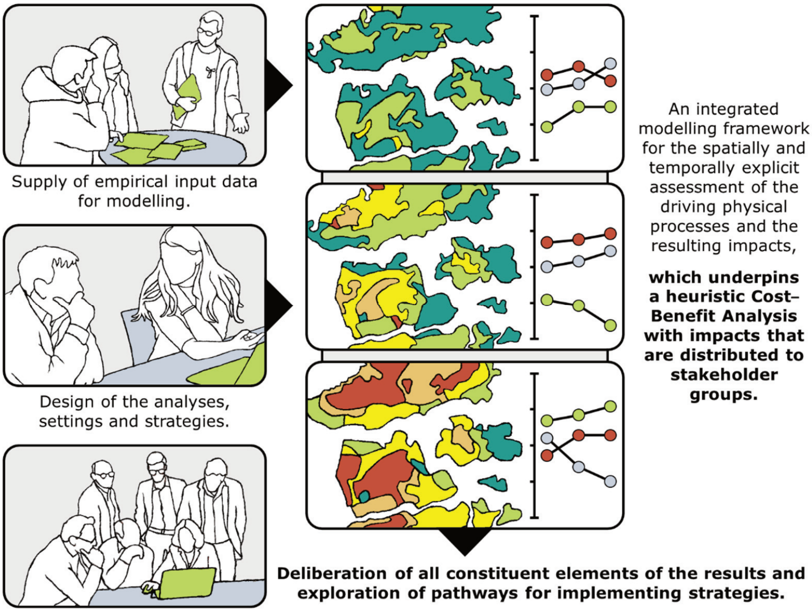

Fig. 1.4. General design of the science–policy interface of step 2. Triangles indicate the direction in which the features feed into each other. New features that are added to the science–policy interface in step 2 are displayed in bold. See Fig. 3.2 for a more detailed design of the decision support tools.

The second step in the development of the SPI adds several new features to the SPI (Fig. 1.4). Regarding the decision support tools, the modelling framework is used to underpin a heuristic CBA. Regarding the participatory processes, all constituent elements of the results are discussed, and pathways for implementing collaborative management strategies are explored. The heuristic CBA approach is derived from Dutch CBA guidelines (Eijgenraam et al., 2000; Faber and Mulder, 2012; Romijn and Renes, 2013), reflecting a symbiosis of CBA and multi–criteria decision analysis with enhanced discursive features, allowing for multiple evaluative endpoints. First, researchers and practitioners collaboratively define water management strategies and the timeframe of the assessments. Second, the integrated modelling framework is used to assess the physical effects of the management strategies. The stakeholders involved in the case study provide empirical economic data that link the physical effects to the ensuing economic effects, which are distributed among the stakeholder groups affected. Third, all constituent elements of the approach are evaluated in participatory workshops. A deliberative web-based tool is used to gauge the opinions of the stakeholders who participate in the meetings regarding the approach and its constituent elements, and to elicit their suggestions for the follow-up of the collaborative management process. The meetings are attended by approximately 240 people, culminating in an exploration of shared interests and pathways for implementing the management strategies.

<u>Step 3: Participatory non-monetary valuations</u>

The third step in the development of the SPI (Chapter 4) elaborates on the combination of decision support tools with the participatory processes of the second step. As the stakeholders who participated in the second step stressed the need for additional evaluative endpoints for non-monetary impacts, the third step considers both monetary and non-monetary valuations, including intersubjective cultural values. Many authors stress the need for such integrated approaches, because these can support the development of a shared understanding and a dialog about plural values (Beria et al., 2012; Daniel et al., 2012; Guerry et al., 2015; van den Belt and Stevens, 2016; Jacobs et al., 2016; Kenter, 2016). Step 3 contributes to our collective understanding of how such approaches can effectively support the collaborative management of SESs. The guiding research question is: *"What is the added value of a heuristic CBA in combination with a participatory non-monetary valuation of cultural ecosystem services for the support of the collaborative management of SESs?"*

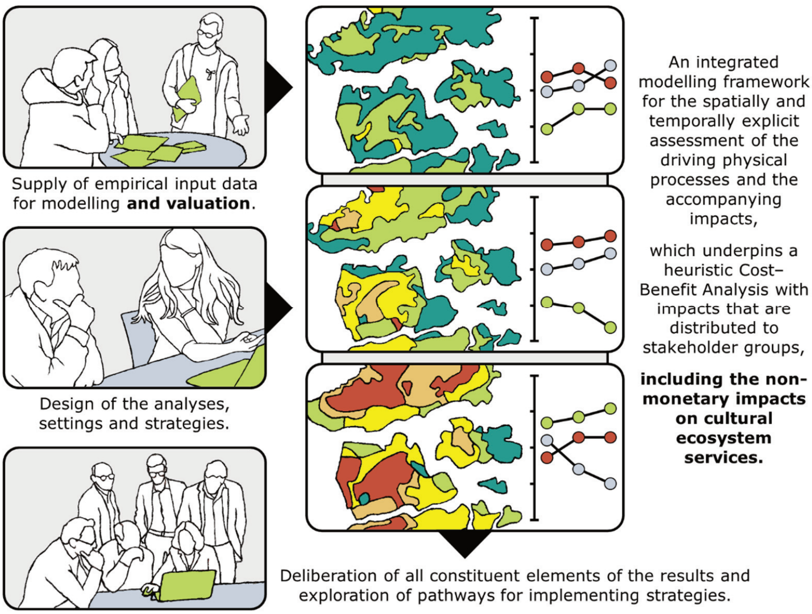

Fig. 1.5. General design of the science–policy interface of step 3. Triangles indicate the direction in which the features feed into each other. New features that are added to the science–policy interface in step 3 are displayed in bold. See Fig. 4.2 for a more detailed design of the science–policy interface.

The third step in the development of the SPI adds several new features to the SPI (Fig. 1.5). Regarding the participatory processes, a new feature is that the stakeholders supply empirical input data for the valuation of cultural ecosystem services. First, several stakeholders jointly design management scenarios and select cultural ecosystem service indicators. Subsequently, the willingness to pay for the scenarios is surveyed, as is the non-monetary valuation of the cultural ecosystem service indicators of 295 stakeholders. Regarding the decision support tools, the survey results allow for an assessment of non-monetary impacts on ecosystem services, which provides additional evaluative endpoints for the heuristic CBA. The combined results of the modelling framework, the CBA and the survey are evaluated in workshops, using a mapping tool that supports deliberation across multiple stakeholder perspectives and value dimensions. In addition, the added value of the approach for the collaborative management of SESs is evaluated.

Step 4: Interactive simulations

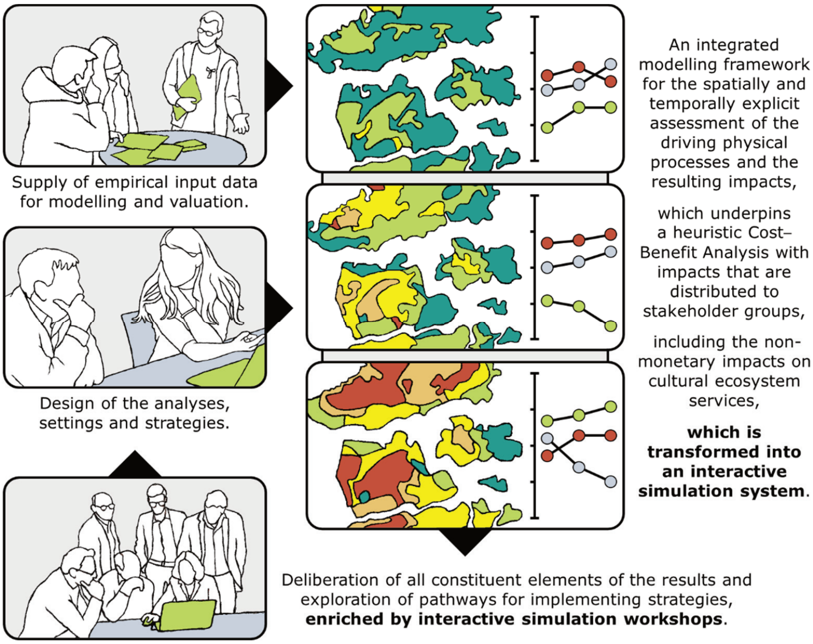

Fig. 1.6. General design of the science–policy interface of step 4. Triangles indicate the direction in which the features feed into each other. New features that are added to the science–policy interface in step 4 are displayed in bold. See Fig. 5.1 for a more detailed design of the interactive simulation system.

The fourth step in the development of the SPI (Chapter 5) presents the final version of the SPI (Fig. 1.6). Regarding the decision support tools, a new feature is the combination of the tools of step 3 with the Tygron Geodesign Platform, an interactive software platform for accurate 3D modelling of spatial development projects (Warmerdam et al., 2006; Bekebrede, 2015). The result is an ISS system called RE:PEAT (an acronym derived from Platform for Evaluating and Anticipating Trends in peatlands), which can be used iteratively to explore the myriad of collaborative management options in peatlands (see Fig. 5.1 for a detailed design of RE:PEAT). Regarding the participatory processes, a new feature is the enrichment of the deliberation processes by interactive simulation workshops. In addition, if desired, the deliberative processes can be followed immediately by an adjustment of the settings and strategies, because RE:PEAT transforms most scenario settings into actions that allow users to influence the simulation.

The guiding research question for step 4 is: *"How can ISSs improve the support of environmental management?"*. Many scholars have reported how collaborative management tasks can be supported by ISSs such as serious games (Bekebrede, 2010; Vervoort et al., 2014; Van der Wal et al., 2016; Voinov et al., 2016; Craven et al., 2017), touch tables (Arciniegas et al., 2013; Eijkelboom and Janssen, 2013; Pelzer et al., 2016), and flood simulation models (Leskens et al., 2014). However, much research is hampered by methodological limitations such as (a) the use of students instead of real-world stakeholders to test their effectiveness, (b) the analysis of only one or a limited number of workshops, and (c) a sole focus on stated opinions of the participants who tested the system, without considering logfiles and/or video recordings of the workshops in which the system was tested. The research that is presented in step 4 overcomes these methodological limitations by applying the ISS in ten workshops which are attended by real-world stakeholders. The applications are evaluated using logfiles, videos, and post-workshop questionnaires. Using the in-depth perspective that is provided by the multiple methodological viewpoints, the added values of ISSs for the support of the management of SESs are discussed.

<u>General discussion and conclusions</u>

The results of the four sequential steps feed into a general discussion and the overall conclusions of this thesis (Chapter 6). In addition, perspectives are presented regarding the implementation of SPIs for the collaborative management of SESs, and the future of the Dutch peatlands.

Chapter 2

An integrated modelling framework to assess long-term impacts of water management strategies steering soil subsidence in peatlands

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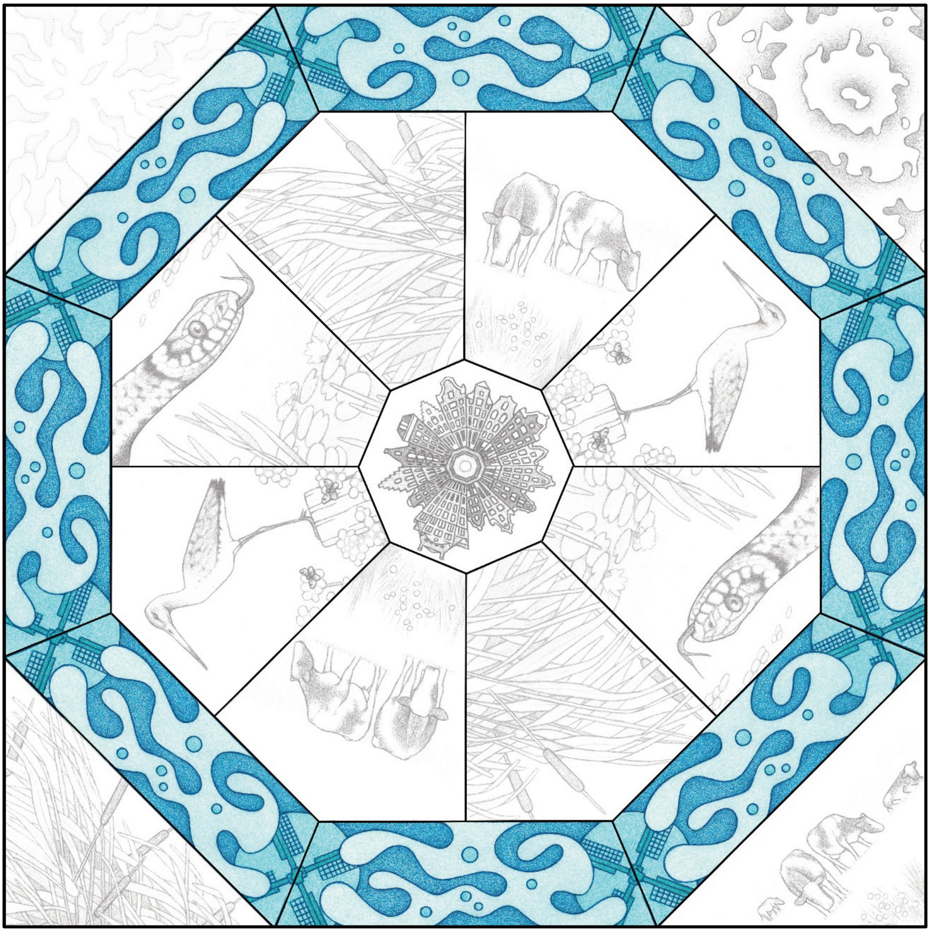

Human exploitation of peatlands requires drainage so that the peat soil is not waterlogged. This causes oxidation, shrinkage and consolidation of peat, leading to soil subsidence. To compensate for soil subsidence, the absolute surface water levels must be lowered periodically, to maintain the same depth relative to ground level. In the figure, the water management is depicted by the octagonal, concentric band.

Abstract

Around the world many peatlands are managed unsustainably. Drainage of the peat causes soil subsidence and a range of negative societal impacts. Integrated strategies are required to ensure more sustainable long-term settings, based on impact assessment models that simulate the interrelated dynamics of water management and soil subsidence, and determine the spatial and temporal range of societal impacts. This paper presents an integrated modelling framework that meets these requirements. We used the framework to assess the impacts of a range of water management strategies in the Dutch peatlands. Average soil subsidence rates were shown to range from 0.6 to 4.5 mm y^{-1}, resulting in marked differences in societal impacts that affect stakeholders unequally. Moreover, the impacts on real estate damage and water system management revealed inverse trends that result in increasingly unbalanced cost–benefit ratios. The generated insights led the regional water authority to change their current water management strategy, preventing unsustainable future developments. We find the results relevant for improving stakeholders' awareness of long-term impacts of management strategies, and making negotiation processes on goals, means, and possible future pathways more transparent.

2.1 Introduction

Unsustainable human exploitation has resulted in loss of peatlands worldwide (Joosten and Clarke, 2002; Bragg and Lindsay, 2003). Human exploitation of peatlands requires drainage so that the peat soil is not waterlogged. This causes oxidation, shrinkage and consolidation of peat, leading to soil subsidence (Schothorst, 1977; Wösten et al., 1997). To compensate for soil subsidence, the absolute surface water levels must be lowered periodically, to maintain the same depth relative to ground level. Although this provides short-term benefits such as increased agricultural production, in the long term it leads to soil subsidence, emission of greenhouse gasses, and loss of biodiversity (Millennium Ecosystem Assessment, 2005; Page et al., 2009; Hooijer et al., 2010). Continuous soil subsidence can cause additional problems such as diminishing agricultural yields and increasing management costs (Verhoeven and Setter, 2010; Querner et al., 2012). All these long-term effects can be diminished by raising the surface water levels and consequently slowing down the soil subsidence rate. However, this will cause the agricultural revenues to diminish as well.

The unsustainable exploitation of peatlands goes on because stakeholder interests are conflicting and consensus on the best management is often lacking. To convert the use of peatlands to a more sustainable mode, a 'wise use' is advocated, i.e. an integrated management strategy that addresses the interests of all stakeholders and ensures benefits for future generations (Joosten and Clarke, 2002). Den Uyl and Wassen (2013) advocate that policy-makers should focus on slowing down peat subsidence and develop a strategy that ensures the required long-term settings for this on a time-scale from decades to a century. They point out that a fair and transparent negotiation process is required on goals, means, and future pathways.

In response to the challenge of a sustainable use of complex social–ecological systems such as peatlands, adaptive management approaches are increasingly put into practice. Although the characteristics of these approaches vary, they are all based on the notion that sustainable management can be supported by a structured process of cooperative learning-by-doing among stakeholder groups (Scarlett, 2013; Den Uyl and Driessen, 2015). However, it is notoriously complex to support a process of ongoing learning and evaluation of the management of social–ecological systems, because many key drivers are uncertain and change nonlinear (Walker et al., 2002), socioeconomic and biophysical processes are strongly interrelated (Pettit and Pullar, 2004; Page et al., 2009; Rawlins and Morris, 2010; Holman et al., 2014), and detailed information is needed to capture heterogeneity and location specific impacts (Van Meijl et al., 2006; Verburg et al., 2008).

Previous assessments of soil subsidence have mainly focused on the physical process of subsidence itself or a limited number of impacts such as agricultural production, water management or greenhouse gas emissions (Schothorst, 1977; Wösten et al., 1997; Hooijer et al., 2010; Verhoeven and Setter, 2010; Querner et al., 2012). However valuable these assessments may be, more integrated assessments are advocated to support a wise use of peatlands (Joosten and Clarke, 2002). Brouns et al. (2015) used GIS applications to make a spatial explicit assessment of water management, soil subsidence and land use in Dutch peatlands and found that these assessments could support an effective change of ideas on adaptation measures. Their findings emphasize the added value of GIS to integrate the spatial and temporal variability of a range of impacts (González et al., 2011). However, a combination of approaches is recommended to obtain more integrated assessments dealing with all the various challenges (Perminova et al., 2016). Lempert et al. (2009) judged adaptive agent-based models useful to explore complex long-term management challenges involving multiple stakeholders, provided they are embedded in a quantitative analytical framework to adequately address biophysical processes. Holman et al. (2014) combined integrative quantitative models with a participatory process of scenario development. This combination of approaches improved the collective understanding of adaptation choices, which therefore could facilitate the development of long-term policies.

The question remains what long-term impact models are useful to support management strategies in peatlands, and how they can be integrated. We took up this challenge by developing a GIS-based integrated modelling framework that enables the interrelated simulation of water management and soil subsidence, and assesses a range of resulting societal impacts. We applied this framework to a peatland area in the Netherlands, and evaluated the added value for exploring the long-term impacts of management strategies in peatlands.

2.2 Methods

Research area

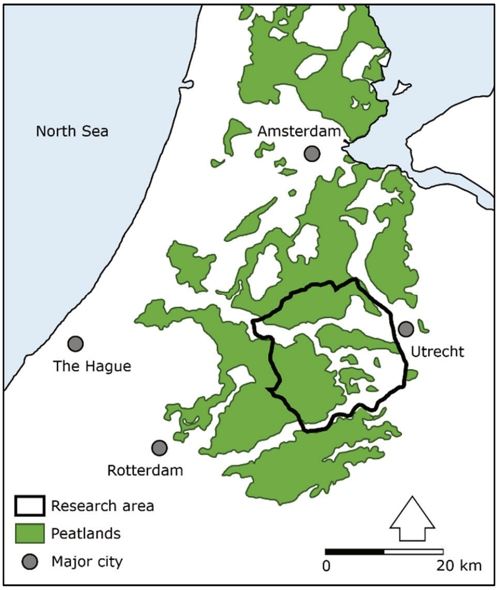

Fig. 2.1. Location of the research area in the western part of the Netherlands.

We focus on a part of the peatlands in the western part of the Netherlands, covering 440 km² (Fig. 2.1). During the Holocene eutrophic wood-sedge peat deposits up to 8 m thickness accumulated when the groundwater table gradually rose with post-glacial sea level rise. The peat deposits are veined with fluvial sand and clay deposits of branches of the river Rhine. During the Middle Ages the natural fens were converted to agricultural fields and meadows. This required drainage to allow oxygen to enter the plant root zone. To achieve this, artificial catchments called polders were created with a dense network encompassing several thousand km of watercourses and hundreds of traditional windmills.

The surface water levels in the watercourses determine the depth of the groundwater table below ground surface, which steers the degree of consolidation and shrinkage of the peat soil, as well as the depth to which oxygen can enter the soil, causing peat oxidation. The surface water levels in the watercourses are therefore the basic steering factor for soil subsidence. Although for centuries the surface water levels remained high, i.e. only shallow drainage was applied, the cumulative soil subsidence over 8–10 centuries nevertheless amounted to approximately 2 m (Schothorst, 1977). From the late 19th century onwards most windmills were replaced by engine driven pumps, which led to lower surface water levels, and hence an increase of the soil subsidence rates. This resulted in current land elevation ranging from +1 m to -2.5 m relative to sea level. Although the peatlands have been subsiding for centuries, their medieval water system and allotment patterns still exists and are acclaimed as valuable Dutch cultural heritage. Nowadays the predominant land uses are dairy farming and urban areas, with approximately 275,000 inhabitants.

<u>Management strategies explored on their impacts</u>

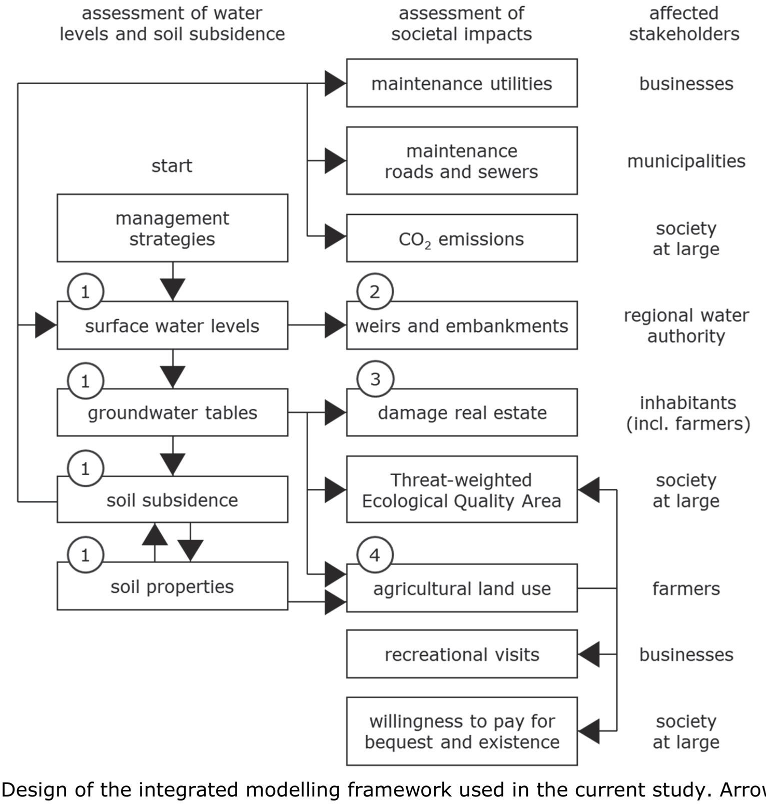

Fig. 2.2. Design of the integrated modelling framework used in the current study. Arrows indicate the sequence of the assessments. Numbered circles indicate the GIS-models we developed.

We used our integrated modelling framework (Fig. 2.2) to compare the impacts of three water management strategies in the research area:

1. Low surface water levels. This management strategy reflects current agricultural land use regardless of future impacts on other interests. In the rural parts of the research area, the surface water levels in the watercourses are maintained at 90 cm below the ground surface, and must be lowered periodically to compensate for the soil subsidence.

In the urban parts of the research area the surface water levels in the watercourses are maintained at the same absolute level throughout time.

2. Current surface water levels. This management strategy reflects current policy, which can be regarded as a compromise between facilitating the current agricultural land use and restricting future soil subsidence. In the rural parts of the research area, the surface water levels in the watercourses are maintained at 30–70 below the ground surface, and must be lowered periodically to compensate for the soil subsidence. In the urban parts of the research area the surface water levels in the watercourses are maintained at the same absolute level throughout time.
3. Progressively higher surface water levels. This management strategy reflects minimizing future soil subsidence, which will negatively affect current agricultural land use. All surface water levels in the watercourses are maintained at the same absolute level throughout time. This implies that, as soil subsidence progresses, the surface water levels will become closer relative to the ground surface leading to increasingly wetter conditions, and lower soil subsidence rates.

We considered a timeframe from 2010 to 2100. Regarding demography and urbanization, the national projections used in the Dutch Delta Programme assume that up till 2050 several diverging scenarios are equally plausible, and developments after 2050 become highly uncertain, especially on a regional scale. Therefore, we assumed the current population and the extent of urban areas would remain unchanged in the research area.

<u>Design of water levels and soil subsidence assessment</u>

In previous research, several equations have been used to assess the long-term impacts of water management strategies steering soil subsidence in peatlands. Most equations consist of (a) a groundwater table component, (b) a soil properties component, and (c) several empirical constants (Van der Meulen et al., 2007; Van den Akker et al., 2008; Zanello et al, 2011; Hoogland et al., 2012). Currently, most soil subsidence assessments in the Netherlands use the equations of Van den Akker et al. (2008), because the required spatial and temporal explicit input data are available, and the empirical constants of those equations apply to all Dutch peatlands (Querner et al., 2012; Brouns et al., 2015). We used the equations of Van den Akker et al. (2008) for our GIS-model for water levels and soil subsidence (model 1 in Fig. 2.2) for these reasons too. See appendix 2A for a more detailed comparison of equations.

For each water management strategy, the GIS-model for water levels and soil subsidence calculates the effects of the surface water levels on the groundwater tables. Subsequently the groundwater tables and the soil properties are used to calculate soil subsidence, which determines to what degree the surface water levels and the soil properties will change in the next time-step.

The GIS-model for water levels and soil subsidence requires initial conditions for the surface water levels of all watercourses, the Average Deepest Groundwater table (ADG), and the Average Highest Groundwater table (AHG), defined as the average of the annual three deepest and highest groundwater tables measured in 14 day intervals for a period of 8 consecutive years. The initial conditions for ADG and AHG are identical for management strategy 2 and 3, and were calculated with a resolution of 25 by 25 m using the operational groundwater model of the regional water authority, which combines the MODFLOW code (McDonald and Harbaugh,1988) and the SIMGRO code (Van Walsum et al., 2007). The groundwater model was constructed using geological properties, measurements of groundwater extractions, and current surface water levels of all watercourses, and was calibrated with a representer-based inverse method (Valstar et al., 2004) using 944 time series of groundwater measurements. The initial conditions for management strategy 1 were derived from the initial conditions of management strategy 2 and 3, by calculating the change in ADG and AHG as a fraction of the change in surface water level relative to the soil surface. The fraction reflects that when shallowly drained soils are waterlogged, excess precipitation does not infiltrate in the soil, but is drained away by surface runoff, diminishing the annual fluctuation in groundwater tables that would occur in more deeply drained soils (Wind, 1987). We assumed a fraction of 67% for our research area.

The GIS-model for water levels and soil subsidence also requires initial conditions for soil texture and strata. The required soil strata were derived from the 3D geological property 'GeoTOP' model (Stafleu et al., 2011) and a soil map (Stouthamer et al., 2008). The 'GeoTOP' model is a voxel model of the upper 30–50 m of the subsurface of the Netherlands, with individual soil properties for each voxel, measuring 100 by 100 m (horizontal) and 0.5 m (vertical). Soil properties for each individual voxel were derived by a stochastic interpolation of almost 500,000 borehole descriptions. For the research area, the lithology of the top 1.2 m was refined by adding data from a 1:25,000 soil map, resulting in voxels measuring 25 by 25 m (horizontal) and 0.3 m (vertical).

Based on the ADG and the soil properties that apply for each time-step, the GIS-model for water levels and soil subsidence calculates soil subsidence using equation [1].

[1] $\Delta S = 23.54 * ADG - 18.34 * CL - 6.68$

ΔS = Rate of soil subsidence (mm y^{-1})
ADG = Average Deepest Groundwater table (m below surface)
CL = Thickness of the clay layer on top of the peat layer (m)

This equation is adapted from the equations by Van den Akker et al. (2008), who used data from literature and measurements of groundwater tables and soil subsidence of 14 parcels at 5 locations in the Netherlands during more than 30 years. This empirical relation thus incorporates all drainage related processes that cause soil subsidence, i.e. oxidation, shrinkage and consolidation, without explicitly assessing their relative contribution to the total soil subsidence

Soil subsidence was calculated over time-steps of five years, during which surface water levels, ADG, ADH and CL were kept constant. The time-step was chosen because it best reflects the average readjustment period of surface water levels in Dutch peatlands. After each time-step, the soil properties are updated, by subtraction of the cumulative soil subsidence over that time-step from the uppermost voxel with peat soil properties. If this results in the disappearance of a peat voxel in between two clay voxels, the CL value for the next time-step is adjusted. Furthermore, for management strategy 1 and 2 the surface water levels, ADG and AHG are lowered with the same rate as the soil subsidence simulated over that time-step. Because in management strategy 3 the surface water levels relative to the ground surface change, for this strategy the ADG and AHG is recalculated as a 67% fraction of the change in surface water levels (similar to how we obtained the initial ADG and AHG conditions for management strategy 1).

Design of societal impact assessment

The output of the water levels and soil subsidence model was used as input in the assessment of societal impacts. We identified two governmental stakeholders, i.e. the regional water authority and municipalities, and three societal stakeholders, i.e. inhabitants, farmers, and businesses. We also included 'society at large', because several non-financial impacts cannot be directly linked to a stakeholder. To assess the impacts on the stakeholders' interests we designed additional GIS-models for weirs and embankments, real estate damage, and agricultural land use (Fig. 2). We used the output of the models, combined with empirical data to assess the impacts on the maintenance of roads, sewers and utilities, the CO_2 emissions, the threat-weighted ecological quality area (T-EQA), the recreational visits, and the willingness to pay (WTP) for bequest and existence values.

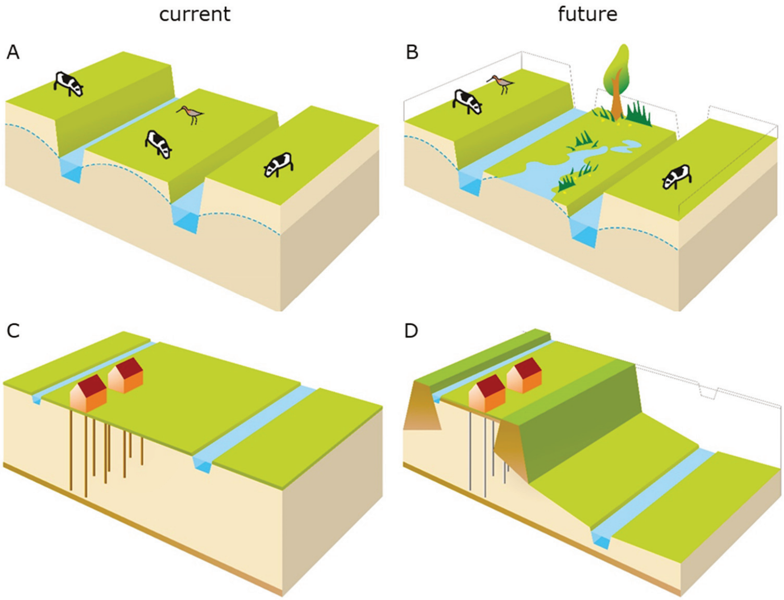

Fig. 2.3. Effects of spatial differences in soil subsidence in Dutch peatlands on depth of the groundwater table and associated possible land use (A and B), and need for additional embankments, to prevent real estate damage (C and D).

The prime interest of the regional water authority is the maintenance of the water system, especially the required weirs and embankments. In the research area, the surface water levels in watercourses are managed in several hundred sub-catchments, each with independently controlled water levels. The management of this complex, predominantly man-made water system currently requires 99 pumping stations, 1,525 weirs and 322 km of embankments. Ongoing soil subsidence causes a further increase in the complexity of the management tasks. To bridge increased differences in elevation, higher embankments and additional weirs are required. Moreover, increased differences in water level between adjacent watercourses will require additional embankments to prevent the watercourses with higher water levels from slumping. In the rural parts of Dutch peatlands, farmsteads and houses are built adjacent to each other in narrow zones parallel to a watercourse (Fig. 2.3C and D). The surface water levels in these watercourses must be kept high, because the house foundations (vertical lines underneath the houses in Fic. 2.3C and D) require high groundwater tables to prevent damage. In contrast, surface water levels in adjacent agricultural fields (right hand side of Fig. 2.3C and D) must be periodically lowered to keep pace with soil subsidence. When the difference in surface water levels between adjacent watercourses exceeds 60 cm, embankments are needed to prevent the watercourse with high surface water levels from slumping (Fig. 2.3D).

The GIS-model for weirs and embankments (model 2 in Fig. 2.2) requires data of the locations of watercourses and sub-catchments, and output from the GIS-model for water levels and soil subsidence regarding surface water levels. The GIS-model assesses the number of weirs required to manage different surface water levels in adjacent sub-catchments. The weirs used in the research area can manage differences in surface water levels up to 60 cm. If the difference in surface water levels exceeds 60 cm, a sequence of consecutive weirs is used to bridge the difference, with each weir bridging a maximum of 60 cm. The GIS-model also assesses the locations where the difference in surface water level between adjacent parallel watercourses exceeds 60 cm. These locations require embankments to prevent the watercourses with high surface water levels from slumping. At each time-step, the calculated soil subsidence determines the raise that is required to maintain the height and breadth of the embankments. Embankment slope angles were set to 1:4.

The prime interest of the municipalities is the maintenance of the public infrastructure, i.e. roads and the sewer system. Maintenance and lifespan of the infrastructure is linearly dependent on the rate of soil subsidence: higher rates give rise to increased frequency of maintenance and shorter lifespans. The impact on the maintenance of roads and sewers was derived by dividing the cumulative soil subsidence at locations of roads and sewers with empirical data on maintenance time intervals. Brick roads need maintenance after 15 cm of soil subsidence, asphalt roads after 10 cm, and sewers after 25 cm.

The prime interest of inhabitants has been attributed to their real estate. Approximately 10,250 houses in the research area have foundations that are not entirely made of concrete and thus are prone to damage (Table 2.1). If groundwater tables drop below a certain threshold wooden foundations oxidize and start to decay, and shallow brickwork foundations lose part of their structural integrity. Depending on the size of the house the resulting damage can amount up to €50,000–200,000 per house.

Table 2.1. Dominant foundation type and threshold for foundation damage per construction period, of houses in Dutch peatlands. [ADG = Average Deepest Groundwater table].

Construction period	Number of houses	Dominant foundation type	Damage threshold [cm fall in ADG since construction]
< 1920	1,730	Shallow brickworks	50
1920–1959	1,670	Wooden poles	20
1960–1974	2,933	Wooden poles topped with 50 cm concrete	70
1975–1989	3,926	Wooden poles topped with 100 cm concrete	120
> 1990	4,007	Concrete poles	none

The GIS-model for real estate damage (model 3 in Fig. 2.2) uses the ADG simulated by the GIS-model for water levels and soil subsidence, in combination with spatial explicit GIS data of the age of the houses. We used empirical data of contractors to establish the construction periods in which different types of foundation were dominant, and assumed that all houses constructed during those periods have the dominant foundation type of that period (Table 2.1). Except for concrete foundation poles, all foundation types are prone to damage if ADG falls below the thresholds mentioned in Table 2.1. For each time-step the real estate damage model indicates which houses are likely to have damaged foundations, by comparing the damage thresholds in Table 2.1 to the calculated fall in ADG since construction. We assumed the fall in ADG between the year of construction of a house and the start of our simulations (2010) is equal to the soil subsidence rate of the management strategy 2 (current surface water levels) in 2010 multiplied by the age of the house in 2010.

The prime interest of farmers is their agricultural production. At present, the rural part of the research area is predominantly used for dairy farming (cows in Fig. 2.3A and B). The most influential parameter for agricultural yield is the groundwater table (dotted lines in Fig. 2.3A and B). If the groundwater table becomes too shallow, crop yield diminishes (depicted by meadow birds that prefer high groundwater tables in Fig. 2.3A and B). When groundwater tables rise too much, profitable dairy farming is no longer possible, and most likely will be replaced by biomass crops such as willow coppice, and reed (depicted by tall grasses and a tree in Fig. 2.3B) that can cope with shallow groundwater tables, but at present are less profitable (Londo et al., 2001; Kuhlman et al., 2013).

The GIS-model for agricultural land use (model 4 in Fig. 2.2) uses the ADG and AHG simulated by the GIS-model for water levels and soil subsidence, and the so-called HELP-tables, to calculate agricultural crop yield reductions. The HELP-tables define relationships between ADG, AHG and crop yields at the field-scale for a range of the most common soil profiles in The Netherlands, with a distinction between crop yield reductions due to wet and dry conditions (De Vos et al., 2006). When crop yield reduction exceeds a certain threshold, we assumed dairy farming to be less profitable than the production of biomass crops, prompting farmers to change the land use. The HELP-tables cannot predict this threshold directly, because these do not take into account adaptations to suboptimal conditions by farmers. However, by cross-analyzing the calculated crop yield reductions with economic data from farms in the research area, we estimated the threshold to be approximately at 40% crop yield reduction. Conditions for biomass crop production remain adequate for a wide range of groundwater tables. Yet, when spring groundwater tables become higher than 15 cm above ground surface, we assumed that biomass crop production becomes unprofitable as well, resulting in land abandonment.

CO_2 emissions, the T-EQA, and the WTP for bequest and existence values are of interest to society at large. The CO_2 emissions were derived using the approach of Van den Akker et al. (2008) to calculate CO_2 emissions from the amount of soil subsidence, average bulk density of peat, organic matter fraction of peat and carbon fraction of organic matter in the top 120 cm of the peat soil. We used the average soil properties that were measured at a monitoring station in the middle of the research area.

The T-EQA is an indicator for ecological values that is commonly used in the Netherlands (Sijtsma et al., 2011). It is derived by multiplying the areas of all natural, semi-natural and agricultural ecosystems with a score for intactness, and a standardized weight factor indicating how much the ecosystem contributes to mean species abundance. We derived the intactness score with empirical relations of soil properties and groundwater tables. We used standardized scores for the Netherlands to derive the score for species abundance, i.e. 0.8 for biomass production, 1.0 for uncultivated land, and 0.4–1.8 for dairy farming, with high crop yields corresponding to low abundance scores, and low crop yields corresponding to high abundance scores.

Bequest and existence values reflect the non-financial benefits people derive from the preservation and existence of nature and cultural heritage. We derived these values with WTP estimates. Using the guidelines for valid benefit transfer of Brouwers and Spanink (1999) and Bos (2007) we transferred the WTP estimates obtained by a survey used for an appraisal of a similar range of land use categories in similar peatlands in the Netherlands, i.e. € 30 y^{-1} for constrained dairy farming (i.e. more than 20% crop yield reduction) which preserves cultural heritage values and breeding meadow birds, € -75 y^{-1} for optimal dairy farming (i.e. no crop yield reduction) which negatively impacts cultural heritage values and breeding meadow birds, and € -90 y^{-1} for biomass crops and uncultivated land with severe negative impacts for cultural heritage values and breeding meadow birds. In accordance with Bateman et al. (2006) we estimated the number of families willing to pay as 47% of the families within a radius of 10 km from the research area.

Several categories of businesses have an interest in Dutch peatlands. The interests of recreational entrepreneurs concern the number of recreational visits to the area, which depends on the agricultural land use, which we simulated with the GIS-model for agricultural land use (model 4 in Fig. 2.2). We used empirical data and estimates for similar peatlands in the Netherlands to estimate 215–230 recreational visits ha^{-1} y^{-1} for dairy farming, and 213 recreational visits ha^{-1} y^{-1} for biomass crops and uncultivated land. The range used for dairy farming reflect opportunities for large-scale activities such as fairs, which decrease as ecological values increase. The lower estimate for biomass crops and uncultivated lands reflect a decrease in the appeal of the landscape, as cultural heritage values and breeding meadow birds are negatively impacted.

The interests of utility businesses concern the maintenance of the commercial utility infrastructure, e.g. gas lines or telephone cables. The impact on the maintenance of the utility infrastructure was derived by dividing the cumulative soil subsidence at locations of utility infrastructure with empirical data on maintenance time intervals. Utility infrastructure needs maintenance after 10 cm of soil subsidence.

2.3 Results

Water levels and soil subsidence

The spatial differences in soil subsidence are pronounced (Fig. 2.4). In the eastern part of the research area and on locations adjacent to the rivers, soils are predominantly composed of sand or clay, intermingled with peat soils topped off with several decimeters of clay. On these locations soil subsidence is either absent, or very slow. In the rest of the area soil is composed of peat or of peat topped off with a relatively thin clay layer. On these locations soil subsidence depends on water management. Management strategy 1 (low surface water levels) results in maximum subsidence rates up to 20 mm y^{-1}., and an average rate for the entire research area of 4.5 mm y^{-1}. In the year 2100 the cumulative soil subsidence amounts to more than 1.0 m in vast parts of the research area. In contrast, the cumulative soil subsidence in management strategy 3 (progressively higher surface water levels) rarely exceeds 0.5 m, with average rates dropping from 2.0 (period 2010–2050) to 0.6 (period 2050–2100) mm y^{-1}. Management strategy 2 (current surface water levels) results in an intermediate amount of soil subsidence.

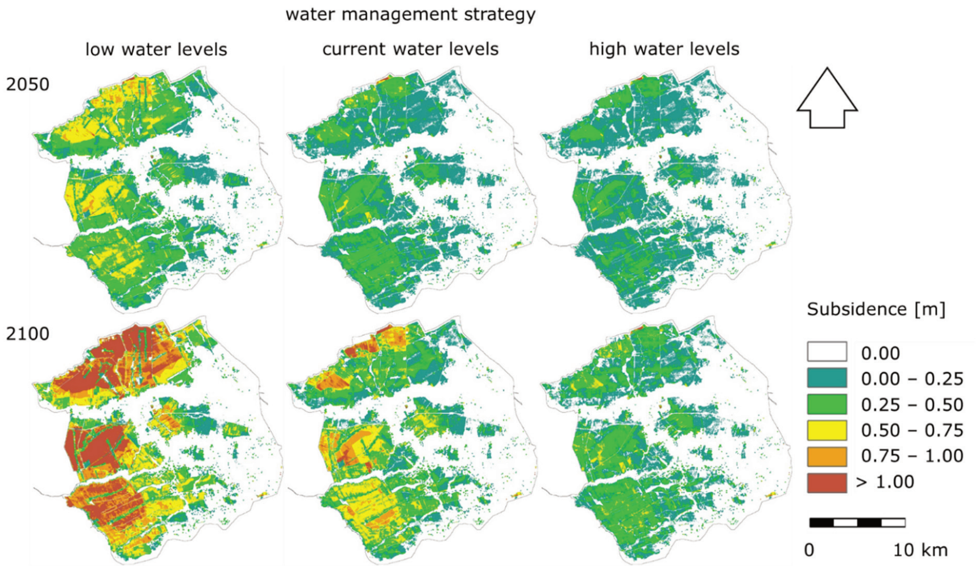

Fig. 2.4. Cumulative soil subsidence since 2010 in the years 2050 and 2100 due to the three water management strategies.

Societal impacts

The societal impacts show distinguished temporal trends (Fig. 2.5) and marked spatial patterns (Fig. 2.6). The maintenance of roads, sewers, and utility infrastructure linearly reflects the cumulative soil subsidence (Fig. 2.5A-C). Management strategy 1 (low surface water levels) results in markedly more maintenance than other management strategies. Relative differences between the management strategies 2 (current surface water levels) and 3 (progressively higher surface water levels) remain small during the first decades, but become more pronounced at the end of the timeframe considered. The cumulative emission of CO_2 (Fig. 2.5D) linearly reflects the cumulative soil subsidence as well. Due to increasingly higher groundwater tables, the yearly CO_2 emission in management strategy 3 (progressively higher surface water levels) decreases from 280.10^6 kg in 2010 to 75.10^6 kg in 2100. The other management strategies result in less pronounced reductions.

The required number of weirs and embankments clearly depends on the management strategy (Fig. 2.5E–G). Soil subsidence increases the required number of weirs to control differences in water levels between adjacent sub-catchments. Moreover, to cope with soil subsidence all embankments need to be heightened and broadened to ensure their stability. For management strategy 3 (progressively higher surface water levels) the current number of weirs, and the current length of embankments is not changed throughout time, and a relatively small increase of the volume of clay needed for embankments. Management strategy 1 (low surface water levels) results in almost 800 additional weirs during the timeframe considered, almost twice the length of embankments, and over a million m^3 of extra clay needed for embankments. Management strategy 2 (current surface water levels) results in intermediate changes.

All management strategies in time result in a decrease of the real estate damage (Fig. 2.5H). The reason is that almost all damage relates to houses built before 1960. The damage threshold for these houses (see Table 2.1) is most frequently breached in the first half of the 21st century. The magnitude of the downward trend is uncertain because it is unclear what the lifespan of the present real estate will be. At present, less than 5% of the houses in the research area is older than 100 years. If a lifespan of 100 years would be assumed, real estate damage would no longer occur from 2050 onwards.

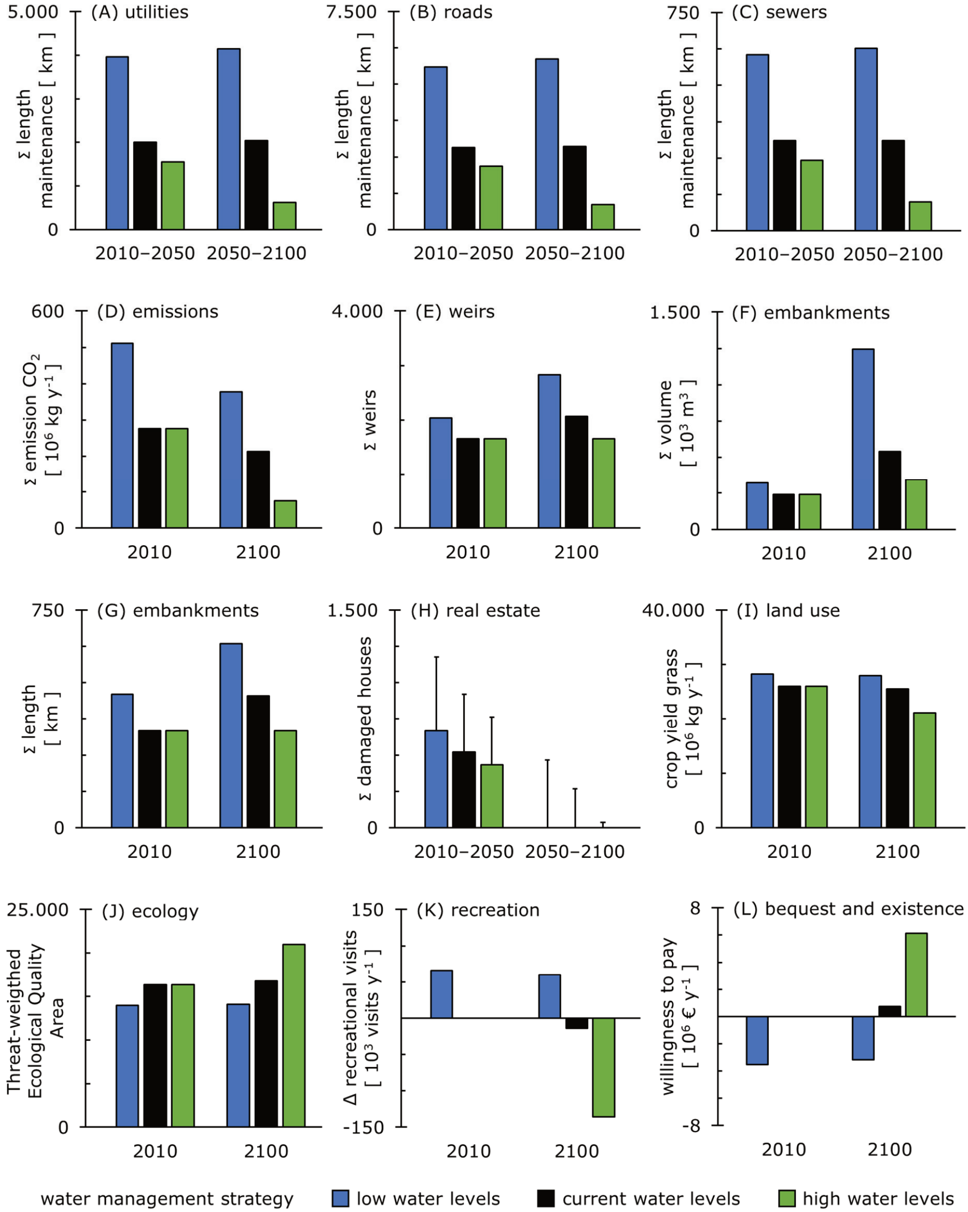

Fig. 2.5. Societal impacts of the three water management strategies. The error bars (H) reflect the range in age expectancy of real estate.

Land use is clearly dependent on the water management and the resulting soil subsidence (Fig. 2.6). In areas with limited soil subsidence (see Fig. 2.4) the land use remains unchanged, while in rural areas with more pronounced soil subsidence, due to progressively higher groundwater tables the land use changes successively from dairy farming to constrained dairy farming (i.e. more than 20% crop yield reduction), biomass crops and ultimately uncultivated land. Management strategy 3 (progressively higher surface water levels) results in pronounced changes in land use. Because the absolute surface water levels remain unchanged and the soil subsides, in time both surface water levels and groundwater tables become higher relative to the ground surface. This constrains the profitability of dairy farming (Fig. 2.5I) and leads to a shift towards biomass crops. The land use in management strategy 1 (low surface water levels) remains to a large extent unchanged. Although soil subsidence is most pronounced in this strategy, the periodically lowering of the surface water levels is sufficient to prevent a major shift in land use. Management strategy 2 (current surface water levels) results in a moderate change in land use.

In the eastern part of the area locations with uncultivated land are present now and will continue to be there (Fig. 2.6). These are caused by anomalous water management since the city of Utrecht is intended to expand there. Therefore, the water management is no longer aimed at optimal facilitation of dairy farming, resulting in a patchwork of relatively wet locations that are unsuitable for agricultural production.

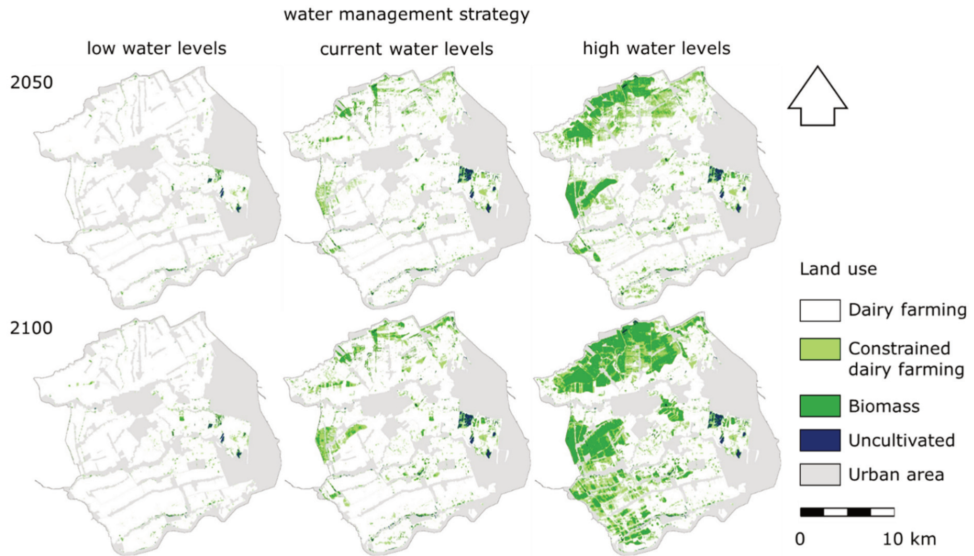

Fig. 2.6. Predicted land use in the years 2050 and 2100 due to the three water management strategies.

T-EQA values (Fig. 2.5J) reflect the land use patterns. Management strategy 1 (low surface water levels) has the lowest T-EQA, because specie abundance is low for dairy farming with high crop yields. Management strategy 3 (progressively higher surface water levels) has the highest T-EQA, because specie abundance is high for dairy farming with low crop yields. The impacts on bequest and existence values (Fig. 2.5L) reveal a similar pattern. The impacts on recreational visits (Fig. 2.5K) are opposite, because opportunities for large-scale recreational activities such as fairs decrease as ecological values increase.

Table 2.2. Impacts on stakeholders' interests of management strategies with low and high water levels, compared to the management strategy with current water levels.

Stakeholder	low water levels	high water levels
inhabitants	negative: more real estate damage	positive: less real estate damage
farmers	positive: higher crop yield	negative: lower crop yield
businesses	negative: more maintenance of utility cables	positive: less maintenance of utility cables
	positive: more recreational visits	negative: less recreational visits
water authority	negative: more weirs and embankments	positive: less weirs and embankments
municipalities	negative: more maintenance of roads and sewers	positive: less maintenance of roads and sewers
society at large	negative: more emissions, lower ecological quality, and lower bequest and existence values	positive: less emissions, higher ecological quality, and higher bequest and existence values

A comparison of the impacts of the management strategies reveals that they affect the stakeholders unequally (Table 2.2). Management strategy 1 (low surface water levels) has a pronounced negative impact on the real estate damage, the maintenance of roads, sewers, and utilities, the required number of weirs and embankments, the emission of CO_2, the ecological quality, and the bequest and existence values. Therefore, this strategy is least desirable for the inhabitants, the municipalities, the utility businesses, the regional water authority, and society in general. Simultaneously, because agricultural conditions remain good, this strategy is most desirable for the farmers and the recreational entrepreneurs. The reverse applies to management strategy 3 (progressively higher surface water levels), which favors the interests of inhabitants, municipalities, utility businesses, regional water authority, and society in general, but negatively impacts the interests of farmers and recreational entrepreneurs.

2.4 Discussion

Wise use of peatlands requires an integrated strategy that addresses the interests of all stakeholders, slows down peat loss and ensures the required long-term settings for this (Joosten and Clarke, 2002; Den Uyl and Wassen, 2013). Our case study illustrates the complexity of this management problem. Management strategies 1 (low surface water levels) and 2 (progressively higher surface water levels) reversely impact the interests of, on the one hand inhabitants, municipalities, utility businesses, regional water authority, and society in general, and on the other hand farmers, agricultural businesses, and recreational entrepreneurs. In response to complex management problems like this, adaptive management approaches are advocated, i.e. structured processes of cooperative learning-by-doing among stakeholder groups. We believe our integrated framework of spatially explicit GIS-models can be of added value to support such processes because it (a) raises awareness of long-term consequences of water management strategies, (b) reveals the associated societal impacts, and (c) allows for a fair and transparent negotiation process on goals, means, and future pathways.

These added values were clearly materialized by the use of our framework by the regional water authority of the research area. In the previous decades, the regional water authority aimed to support the current agricultural land use by managing the surface water levels according to management strategy 2. Simultaneously, they aimed to prevent ensuing damage to the foundations of the real estate of inhabitants. To achieve this, they constructed many sub-catchments with raised surface water levels (Fig. 2.3D). Our research raised awareness that continuation of this policy in the long run would require many additional embankments (Fig. 2.5G), while simultaneously the prevented real estate damage diminishes (Fig. 2.5H). The regional water authority used our framework to assess further impacts up to the end of the 22th century, and used the results as input for a Cost–Benefit Analysis. They found that in the timeframe considered their cumulative maintenance costs of the sub-catchments amounts to €550–630 million, whereas the cumulative prevented damage to the real estate of inhabitants amounts to €120–220 million. So, benefits clearly would not outweigh the costs. Moreover, their annual maintenance costs increase with 30%, whereas the number of damage-prone houses decrease with 90%. This increased insight of long-term societal impacts led them to change their water management strategy. Henceforth they will focus on prevention of unequal soil subsidence rates and large differences in terrain elevation. Consequently, the embankment of sub-catchments with high surface water levels will no longer be required, and high maintenance costs will be prevented. Inhabitants that suffer unacceptably from this change in strategy will be financially compensated, which costs are of a far smaller order than the costs of continuing of the current strategy.

We envision our modelling framework can support policy processes in other peatlands in a similar way. The models quantify an integrated set of long-term impacts of management strategies, and consider the spatial and temporal dynamics of soil subsidence. This results in more detailed information than tools that merely extrapolate an assessment for the current situation, or focus on one specific impact. The assessed societal impacts can be cross-analyzed to enrich the understanding of the peatland dynamics, including insights in inverse trends that would not be revealed by less sophisticated frameworks. Because these insights improve stakeholders' awareness of the long-term impacts of their actions, it can be a strong incentive to focus management strategies on long-term impacts instead of short-term problems, thus avoiding short-term actions that result in increasingly unbalanced cost–benefit ratios, which in the long-term are difficult to amend, but in the current situation can still be avoided.

Our framework also enables evaluation of the equity of different management strategies, because it reveals which stakeholders are unequally exposed to the consequences of management strategies. Therefore, the modelling framework can make negotiation processes on goals, means, and future pathways more transparent, which will support the stakeholders in their adoption of more 'wise' management strategies. The relevance of this has been pointed out by Runhaar (2016), who illustrates that the impact of analytical integration tools on policy-making and planning is usually modest, precisely because they lack to provide insight in socioeconomic consequences, or fail to deal with controversies and conflicting interests. We believe our framework can have a more profound impact, because it addresses these issues better than less integrated or solely analytical tools. Moreover, the use of our framework by the regional water authority showed that it is well suited for a combination with social Cost–Benefit Analysis. This will further strengthen the insight into socio-economic consequences and support a fair and transparent negotiation process.

We performed a sensitivity analysis of the soil subsidence assessment to check how sensitive soil subsidence is for assumptions in parametrization. Soil subsidence appeared to be moderately sensitive to uncertainty in climate change parameters (temperature and rainfall) primarily caused by sensitivity to changes in the ADG-constant. The impact of changes in ADG throughout time is limited, because the dense network of watercourses in Dutch peatlands limits the impact of climate change on the ADG to several cm. The impact of a changed CL-constant is limited as well (see appendix 2A). From this we may conclude that our modeling framework is quite robust for the most important assumptions we made. Still, we recommend future users of our framework to be explicitly aware of the implications of this uncertainty while applying the developed framework. We advise the developed framework not to be used for comparing the effects of management strategies that only differ slightly from each other in water levels (some cm).

2.5 Conclusion

We developed a GIS-based integrated framework that considers the interrelated dynamics of water management and soil subsidence, and assesses a range of resulting long-term societal impacts. We applied the framework to a part of the Dutch peatlands and considered three water management strategies, with average soil subsidence rates ranging from 0.6 to 4.5 mm y^{-1}. We found these strategies result in marked spatial patterns and distinguished temporal trends that affect stakeholders unequally. The improved understanding of long-term societal impacts led the regional water authority to change their current water management strategy, preventing unsustainable outcomes in the future.

The added value of our integrated framework for exploring the long-term impacts of management strategies in peatlands is that it:

- improves awareness of long-term impacts of management strategies, by considering the spatial and temporal dynamics of soil subsidence;
- quantifies a range of societal impacts, that can be cross-analyzed to enrich our understanding of the peatland dynamics, and can be a strong incentive to focus management strategies on long-term impacts instead of short-term problems;
- reveals which stakeholders are unequally exposed to the consequences of management strategies, which can make negotiation processes on goals, means, and future pathways more transparent.

Acknowledgements

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Appendix 2A. Limitations of the soil subsidence assessment

Linear soil subsidence equation

The reliability of the soil subsidence model is limited by the linear character of the soil subsidence relation in equation [1]. When fibric peat soil is drained for the first time, subsidence rates are initially high, but eventually become lower (Schothorst, 1977; Wösten et al., 1997; Hooijer et al., 2012). This is because in the first years consolidation and shrinkage of the soil predominate, whereas in a later stage the subsidence is predominantly caused by oxidation. In Dutch peatlands, soils have been drained since the Middle Ages, which resulted in a partly decomposed, mesic peat soil, on top of pristine fibric peat. The mesic top soil is gradually oxidized, which is a relatively slow process. Equation [1] was derived from observations of the cumulative soil subsidence caused by the continuous process of periodically lowering surface water levels to compensate for the soil subsidence (Van den Akker et al., 2008). These observations mainly reflect the gradual oxidization of the mesic top soil, but also include the periodically non-linear consolidation and shrinkage of small layers of fibric peat that become drained for the first time after the surface water levels are lowered. Hence, equation [1] reflects the reality in Dutch peatlands and other peatlands that have been managed for several decades in a similar way. It will however underestimate soil subsidence in peatlands that are drained for the first time, or experience a marked increase in drainage.

Empiric equation for current Dutch climate conditions

Equation [1] is only valid for climatic conditions similar to those in the Netherlands during the previous decades. In warmer regions, similar hydrological conditions may result in much higher soil subsidence rates due to increased peat oxidation under higher temperatures (Wösten et al., 1997). Therefore, in regions with other climate conditions, or in similar regions where climate conditions change the ADG-constant in equation [1], and the ADG input must be adapted. We assessed the sensitivity of the results to these adaptations with two model sensitivity runs: (1) changes in the ADG-constant in equation [1] throughout time, and (2) changes in the ADG throughout time.

The empirical ADG-constant in equation [1] is defined by the temperatures in the Netherlands during the previous decades. We presumed that these temperatures would remain unchanged. However, if the temperature would rise, the microbes that oxidize peat would become more active (Tate, 1987), resulting in higher soil subsidence rates. The first sensitivity run analyzes the impact of higher temperatures by gradually adjusting the ADG-constant in equation [1] from 25.16 (simulation period 2010–2025) to 31.04 (simulation period 2075–2100), which reflects an increase in biological activity due to a regional projection of 2°C global temperature rise (Van den Hurk et al., 2006), assuming average soil properties of Dutch peatlands, and sufficient oxygen availability for optimal microbial activity throughout all soil layers. The second sensitivity run analyzes the impact of changes in precipitation and evaporation. We considered a regional projection of climate change that assumes that in 2050 the average rainfall in summer will decrease by 19%, and the annual potential evapotranspiration in summer will increase by 15% (Van den Hurk et al., 2006). We forced this regional climate change projection on the groundwater model we used to obtain the ADG input, and calculated the change in ADG for each time-step. We then added these changes to the ADG simulated with the GIS-model of water levels and soil subsidence after each time-step.

We compared the model sensitivity runs with the default run for management strategy 2 (current water levels). The model sensitivity runs revealed that the results are moderately sensitive to uncertainty caused by climate change (Table 2A1). The sensitivity to uncertain climate changes is primarily caused by the sensitivity to changes in the ADG-constant. The impact of changes in the ADG throughout time is limited, because the dense network of watercourses in Dutch peatlands limits the impact of climate change on the ADG to several cm.

Table 2A.1. Sensitivity runs soil subsidence assessment. The difference with the default scenario is given in parentheses. [ADG = Average Deepest Groundwater table, CL = thickness Clay Layer]

Scenario	Average soil subsidence [m]	
	2010–2050	2010–2100
Strategy 2. Default settings	0,11	0,22
Strategy 2. Changes in ADG-constant throughout time	0,15 (44%)	0,37 (80%)
Strategy 2. Changes in ADG throughout time	0,11 (1%)	0,23 (5%)
Strategy 2. Changed CL-constant	0,16 (57%)	0,32 (57%)

Querner et al. (2012) assessed that for a part of our research area without a clay layer, with surface water levels somewhat lower than our management strategy 2 (current surface water levels), in 2050 the same regional projection of climate change we considered, increases the soil subsidence rate with 68%. This result is similar to the average results of our sensitivity run with changes in the ADG-constant through time (Table 2A.1). However, because we considered a more diverse research area, with somewhat higher surface water levels, and vast parts with clay layers, if both studies would have assessed the climate change projection in a similar way, our results would be expected to reveal smaller impacts. Regardless of which study overestimated or underestimated the impacts, these results reveal that assessment methods for soil subsidence are sensitive to uncertainty caused by climate change. Future users of our framework should explicitly be aware of the implications of this uncertainty.

<u>No compaction</u>

Another simplification of our soil subsidence model is that it only assesses soil subsidence caused by drainage. Compaction caused by added weight, e.g. of the materials used to raise roads and construction sites, is not incorporated, which will lead to an underestimation of soil subsidence at raised locations.

<u>Uncertain empiric data</u>

Equation [1] was derived by combining an empirical relation for soil without a clay layer, and an empirical relation for soils with a clay layer (Van den Akker et al., 2008). We assumed an average thickness of 0.2 m clay at the locations used to obtain the relation, which resulted in a CL-constant of 18.34. We assessed the sensitivity of the results to this assumption by a third sensitivity run that analyzes the impact of a CL-constant of 12.63, which reflects an assumed average thickness of 0.3 m at the locations used to obtain the empirical relation. The model sensitivity run revealed that the results are moderately sensitive to the uncertainty of the CL-constant (Table 2A1). The uncertainty of empiric input data of clay layer thickness will have a similar impact. Future users of our framework should explicitly be aware of the implications of this uncertainty.

Comparison to other soil subsidence equations

In previous research, several equations have been used to assess the long-term impacts of water management strategies steering soil subsidence in peatlands. We consider the equations of Van den Akker et al. (2008) best fitted for our GIS-model for water levels and soil subsidence (model 1 in Fig. 2.2) for three reasons: (1) the equations consider all drainage related processes and relate them to both groundwater tables and soil properties, (2) the required input data for the equations are spatial and temporal explicitly available, (3) the empirical constants of the equations apply to our research area. In the remainder of this section, we review three alternative soil subsidence equations and point out why we prefer to use the equations of Van den Akker et al. (2008) in our approach.

Van der Meulen et al. (2007) used equation [A1] to assess soil subsidence in all Dutch peatlands. They used an arbitrary value of 50 cm for h_{dry} and reported an empirically obtained value of 15 mm per meter of unsaturated soil per year for V_{ox}, without reporting how they obtained this value. For a scenario that somewhat resembles a mix of our management strategies 1 (low water levels) and 2 (current water levels), they calculated cumulative soil subsidence in 100 years of more than 1 meter, whereas for a scenario that resembles our management strategy 3 (progressively higher water levels) they calculated much lower values. Their results are in line with our assessment, but lack the spatial explicit accuracy of our results.

[A1] $\Delta h_t = h_{dry} * (1 - \exp(-V_{ox} * \Delta t))$

Δh_t = Layer thickness reduction at time t (m)
h_{dry} = Unsaturated zone thickness (m)
V_{ox} = Empirically obtained peat oxidation rate (t^{-1})
Δt = Oxidation time (t)

For our research area, the h_{dry} value can be improved, by using spatial explicit assessments made with the operational groundwater model of the regional water authority (see section 2.2). However, it is unclear if h_{dry} reflects average groundwater tables of averaged deepest groundwater tables. Moreover, because it is not reported how the empirical V_{ox} is obtained, it is unclear in which settings the equation is valid, and to what extent it includes other drainage related soil subsidence processes as well. Furthermore, the equation does not use soil properties as input. For these reasons, we consider the equations of Van den Akker et al. (2008) better suited to assess soil subsidence in our research area.

Zanello et al. (2011) used equations [A2] and [A3] to assess soil subsidence in a peatland near Venice. As input data, they used survey data of soil properties and four-year time series of elevation, soil temperature, and groundwater table. First, they used a numerical model to compute the reversible dynamics of swelling and shrinking of the peat soil, which they filtered form the measured time series. Then, they assumed estimations of k and T_0 from research in the Florida Everglades were valid for their research area as well, and calibrated an empirical relationship relating soil subsidence to soil temperature and drainage depth. Although the Venice coastland is a somewhat different setting than Dutch peatlands, it is noteworthy that for a scenario similar to our management strategy 2 (current water levels), they calculated a cumulative soil subsidence in 50 years of approximately 25 cm. This result matches our results very well.

The approach of Zanello et al. (2011) uses $(E_{u,t} / E_{u,t0})$ to take into account the impact of ploughing on the availability of carbon matter for oxidation. In Dutch peatlands, which are used for dairy farming and are not ploughed, this part of the equation is irrelevant. The other part seems better fitted to assess the temperature dependent oxidation process than the equations of Van den Akker et al. (2008), but does not consider that the consolidation process is not temperature dependent. Arguably, an addition of a fitting parameter that is not temperature dependent might improve the performance. However, because the required input parameters were not available for our research area, we were not able to use the approach of Zanello et al. (2011) and test this hypothesis.

[A2] $s(T,h) = (a + (b * h)) * \exp(k * (T - T_0)) * (E_{u,t} / E_{u,t0})$

s(T,h) = Biochemical subsidence rate at temperature *T* and depth of the groundwater table *h* (mm y^{-1})
a = Fitting parameter (mm y^{-1})
b = Fitting parameter (y^{-1})
h = Annual average depth of the groundwater table (mm)
k = Reaction rate constant (-)
T = Annual average soil temperature at 0.1 m depth (°C)
T_0 = Threshold soil temperature above which the biochemical reaction is active (°C)
$E_{u,t}$ = Carbon content at thickness of organic upper soil *t* (-)
$E_{u,t0}$ = Carbon content at initial thickness of organic upper soil *t* (-)

[A3] $E_{u,t} = E_{u,t0} - (1 - ((\rho_l * E_l) / \rho_u)) * ((t_0 - t) / t^*)$

ρ_l = Bulk density of organic lower soil *l* (kg m^{-3})
E_l = Carbon content of organic lower soil (-)
ρ_u = Bulk density of organic upper soil *l* (kg m^{-3})
t^* = Ploughing depth (m)

Hoogland et al., (2012) used equations [A4] and [A5] to assess soil subsidence in another part of the Dutch peatlands. As input data, they used historic and recent survey data of elevation, soil properties, and water levels. Assuming D equal to 0,8 m, they calibrated the K and C coefficient by iterative calculus with a time step of one year from the historic survey year onwards, continuously updating soil properties and water levels. They calculated an average soil subsidence of 5.3 mm y^{-1} for their research area. This is slightly higher than the results for our management strategy 2 (current water levels), but considering that our research area contains a larger part without peat soils, their results match ours very well.

The approach of Hoogland et al. (2012) uses the approximation $\{E(s,t) - W(s,t)\}$ and the assumption D=0.8 m to take into account the groundwater tables. For our research area, these input data can be improved, by using assessments made with the operational groundwater model of the regional water authority (see section 2.2). However, our research area is more than 22 times larger than the research area of Hoogland et al. (2012), with adequate spatial explicit historic input data only partial available. Therefore, the approach of Hoogland et al. (2012) is not suited for our research area. Moreover, it is unclear how the observed soil subsidence in surveys can be adequately attributed to on the one hand oxidation (the *K* coefficient), and on the other hand other processes such as consolidation and shrinkage (the *C* coefficient). Arguably, the equations of Van den Akker et al. (2008) might be more fitting to explain the observed soil subsidence, because they consider all drainage related soil subsidence processes combined.

[A4] $dE(s,t)/dt = -K * fO(s,t) * \{E(s,t) - W(s,t)\} - C$

dE(s,t)/dt = Rate of soil subsidence at location *s* (mm y^{-1})
E(s,t) = Surface elevation at location *s* and time *t* (mm Dutch Ordnance Level)
K = Fraction of the peat thickness oxidizing each year (y^{-1})
fO(s,t) = Fraction of the upper part of a soil profile where sufficient oxygen is available for oxidation (-)
W(s,t) = Surface water level at location *s* and time *t* (mm Dutch Ordnance Level)
C = Subsidence rate due to other processes than oxidation (mm y^{-1})

[A5] $fO(s,t) = P(s,t) / D$ if $P(s,t) \leq D$
$fO(s,t) = 1$, if $P(s,t) > D$

P(s,t) = Total thickness of the peat layers in the upper 1.2 m of a soil profile at location *s* and time *t* (mm)
D = Maximum depth where sufficient aeration occurs, approximated by the average deepest groundwater table (mm)

Chapter 3

Supporting collaborative policy processes with a multi-criteria discussion of costs and benefits: the case of soil subsidence in Dutch peatlands

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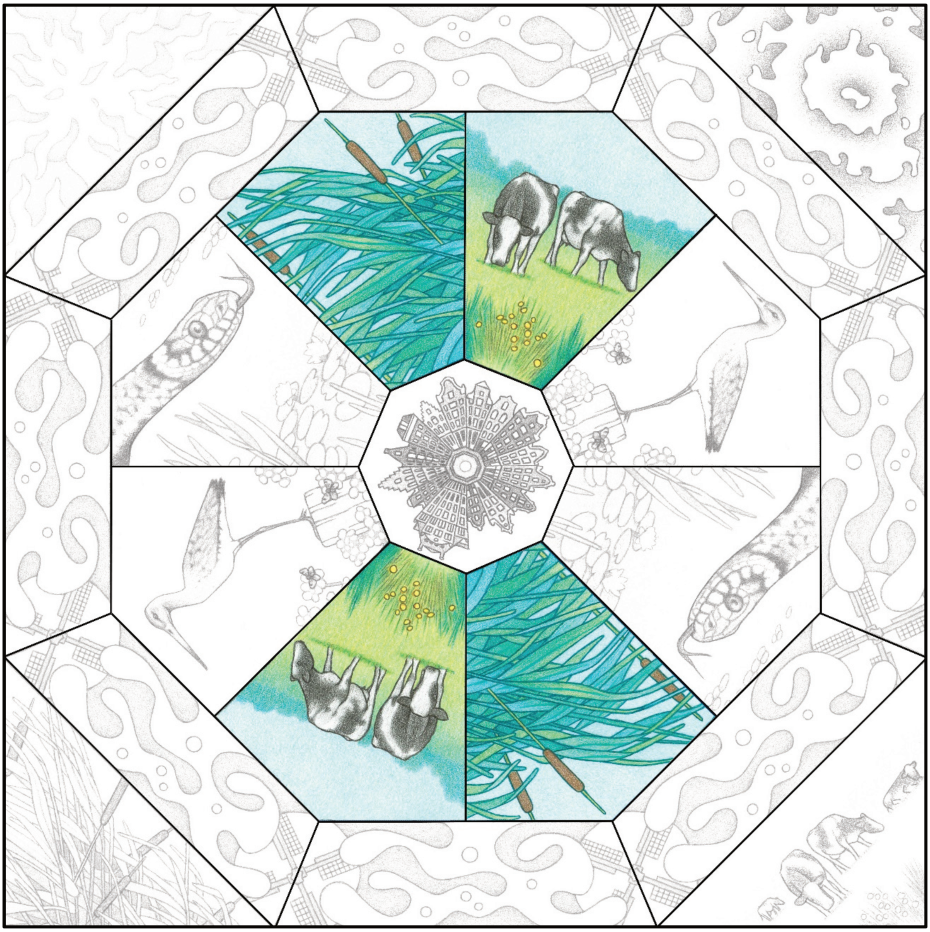

Currently, the rural part of the Dutch peatlands is predominantly used for dairy farming. In the figure, this is depicted by the polygons with the grazing cows. To slow down soil subsidence, the groundwater table must be raised. Most likely, this will result in the replacement of dairy farming by crops which can cope with shallow groundwater tables. In the figure, this is depicted by the polygons with reed and bulrushes.

Abstract

Collaborative policy processes are increasingly advocated to resolve management problems of social–ecological systems. To elucidate which approaches work in diverse situations, this paper demonstrates the added value of Cost–Benefit Analysis in combination with a deliberative tool as a support system a collaborative policy process in Dutch peatlands. We used quantitative models to assess the spatial and temporal physical effects of three water management strategies steering soil subsidence and land use. The stakeholders involved in the case study provided empirical economic data to link the physical effects to the ensuing economic effects, which we distributed among the stakeholder groups affected. The case study aimed for an intersubjective assessment of strategies for water management and land use planning. We therefore enhanced the discursive features of Cost–Benefit Analysis, focusing on knowledge exchange and the evaluation of equitable tradeoffs. The stakeholders participating in our case study appreciated the approach´s comprehensive assessments, and the ensuing multicriteria discussion of the costs and benefits. We believe this result can be attributed to (a) the clear, participatory design of the Cost–Benefit Analysis process, (b) a comprehensive presentation of the constituent elements of the Cost–Benefit Analysis result, and (c) the abundant opportunities to deliberate the results. We discuss how our approach can increase stakeholders' capacity to understand the complexities of social–ecological systems and their ability to explore the potentialities of these systems.

3.1 Introduction

The Millennium Ecosystem Assessment (2005) clearly demonstrated that most of the valuable services ecosystems provide to society are degrading or are being used unsustainably. There are no panaceas for achieving a more sustainable management of social–ecological systems, because interventions often cause nonlinear changes in a complex set of interrelated environmental, political, and economic variables across multiple spatial and temporal scales (Ostrom 2007; 2009). In response to this complexity and unpredictability, adaptive management approaches have emerged that aim to increase the resilience of social–ecological systems through a structured and iterative learning-by-doing strategy (Den Uyl and Driessen 2015). Early versions of adaptive management approaches tended to focus on enhancing the scientific knowledge of the ecosystem being managed. Because the knowledge generated was frequently not successfully linked to management, more iterative approaches that allowed stakeholders to collaborate were designed (Scarlett 2013). The benefits of stakeholder collaboration are legion and can be derived from (a) normative ideas and principles, e.g., the enhancement of democratic capacity or deliberation among participants, (b) a substantive rationale to improve the quality of decisions, and (c) an instrumental underpinning to generate legitimacy or resolve conflict (Glucker et al. 2013).

Although collaborative adaptive management approaches are credited with great potential to improve the management of social–ecological systems, they prove difficult to put into practice. To improve this predicament, social learning processes are advocated, aimed at "learning together to manage together" (Pahl-Wostl et al. 2007; Monroe et al. 2013). To achieve mutual understanding, Van de Riet (2003) points out that the viewpoints of researchers and practitioners must be carefully balanced. Too much focus on researchers' views may produce only "superfluous knowledge", i.e., knowledge that is scientifically valid, but irrelevant to the management problem. On the other hand, too much focus on practitioners' views may result in "negotiated nonsense", i.e., knowledge that is supported by stakeholders but is scientifically invalid.

To reconcile the viewpoints of researchers and practitioners, the integration of analytical and deliberative tools seems to be a prerequisite. For instance, Goosen and Vellinga (2004) promote collaborative planning platforms that include support tools for negotiation and mediation, as well as tools for the assessment of the costs and benefits of the stakeholders involved. Holman et al. (2016) found that the integration of participatory scenario development and quantitative modeling can facilitate dialog among stakeholders and a better understanding of the impacts of management choices. Chaudhury et al. (2013) discuss how participatory scenario analysis can provide the legitimacy needed to bridge disciplinary boundaries and point out that quantification of the scenarios is needed to address the credibility and salience of the knowledge. Quantification of participatory scenarios is especially important if the goal of the process is to make concrete management decisions (Bohunovsky et al. 2011).

Given the context-dependency of most management problems of social–ecological systems, it has been suggested that instead of trying to conjure up a one-size-fits-all solution, more empirical insights from projects should be captured and disseminated, to illustrate which approaches work in diverse situations (McNie 2007; Beratan 2014). Therefore, this paper aims to contribute to this collective understanding by demonstrating how quantitative modeling, Cost–Benefit Analysis (CBA), and a web-based discussion tool were employed to support a collaborative policy process in Dutch peatlands. Some scholars believe that the combination of CBA and deliberative tools has high potential to support collaborative policy processes (De Jong and Geerlings 2003; Turner 2007; Brown and Ryan 2011; Beria et al. 2012). Yet, case studies that demonstrate the added value of such combinations remain underexposed in the scientific literature. This paper aims to fill this knowledge gap.

3.2 Background

<u>Cost–Benefit Analysis as a heuristic aid</u>

CBA has proven its worth for project planning and policy analysis for many decades, with methodological origins going as far back as the definition of benefits and costs by the French economist and engineer Jules Dupuit in the mid-19th century and the stipulation of the principle that the benefits of an investment should exceed the costs (Navrud and Pruckner 1997). Although overall societal wellbeing is improved whenever this principle is applied, this nevertheless implies that those who bear the costs will be worse off. During the 1930s and 1940s the works of Kaldor and Hicks justified this benefit-costs principle by stating that societal wellbeing is improved whenever the gainers can compensate the losers, regardless of whether the compensation occurs (Pearce 1998).

Changes in wellbeing are assessed by comparing the financial and non-financial effects of a measure with the effects of a "business as usual" scenario in which the current policy remains unchanged. Financial effects are fully captured in commercial markets and can be derived from the costs of consumed goods and services and their Net Value Added (NVA) of production, i.e., the sum of producers' income, interest, depreciation, and paid labor. Non-financial effects are not fully captured in commercial markets and require other valuation techniques. In recent decades, the valuation techniques used in CBA have gradually improved, resulting in CBAs that encompass the financial and non-financial economic values of a wide range of ecosystem services (Costanza et al. 1997; Turner et al. 2000; Robbins and Daniels 2012).

The broad scope and the uniform monetary evaluation make CBA potentially a suitable tool to address the complexity of social–ecological systems. However, previous CBAs have encountered a variety of process-related issues that diminish its usefulness as a support tool for collaborative processes. Turner (2007) discusses how the use of CBA as a "decision rule", i.e., the a priori identification of the optimal cost–benefit ratio of project alternatives, often conflicts with the iterative manner of consensus building in real-world policy processes. He suggests that a better match for these processes is the use of CBA as a "heuristic aid", i.e., a complementary component in a decision support system that aims for an intersubjective assessment of the preferred project alternative. Furthermore, both Beukers et al. (2012) and Mouter et al. (2013) found that CBA practitioners perceived misunderstandings and inadequate communication between planners and economist as a core problem in CBA processes. This predicament appears related to opposing views among the CBA practitioners on how CBA should be used. As a result, debates tend to focus on other issues than the management problem at hand, e.g., the limitations of CBA methodology, or the value assigned to CBA in the decision-making process. In addition, if some practitioners are insufficiently aware of CBA methodology, these communication deficits may even result in mistrust, if practitioners believe their values are deliberately disregarded, and the knowledge obtained by the CBA is used strategically.

Remarkably, the CBA practitioners that perceived the processes-related issues still believed CBA should be used in the appraisal process of a project, because it provides valuable information about the usefulness, necessity and design of a project (Mouter et al. 2013). However, to maximize the impact of these advantages, the process-related issues must be dealt with. The suggested solution by some scholars is that CBA should refrain from presenting final verdicts based on decision rules but should instead be used as a method for collecting, organizing, and discussing information relevant to interactive policy making, embedding the analytic analyses in deliberative processes aimed at revealing preferences and settling arguments (De Jong and Geerlings 2003; Robinson and Hammitt 2011). To achieve this, many authors propose a combination of CBA and Multi-Criteria Decision Analysis (MCDA), either by complementing CBA with a MCDA of non-financial values, or by using CBA as one component of a wider MCDA (Turner 2007; Brown and Ryan 2011; Beria et al. 2012).

<u>Water management and land use planning in Dutch peatlands</u>

In the research area (Fig. 3.1) the predominant land uses are dairy farming and built-up areas, and there are some small marshland nature reserves. The area lies in the delta of the river Rhine; its elevation ranges from 1 m above to 2.5 m below sea level. This low elevation requires manipulation of the drainage to prevent the soil from becoming waterlogged. To achieve this, during the Middle Ages artificial catchments called polders were created, with a dense network of several thousand km of watercourses. At present the drainage base levels of the watercourses are maintained at 30–70 cm below ground. Drainage causes the peat soil to oxidize, shrink, and compact, which in the research area results in average soil subsidence rates of 5–10 $mm.y^{-1}$. To compensate for this soil subsidence, the drainage base levels must be lowered periodically, so they remain at the same depth relative to ground level. The soil subsidence results in high management costs for roads, sewers, and the waterways. Furthermore, real estate foundations are prone to damage, and the peat soils emit greenhouse gases and nutrients as they subside. All these effects can be diminished by setting higher drainage base levels, which would result in higher groundwater tables and therefore reduce the soil subsidence rates. However, the revenues of dairy farmers would fall.

For many years, the long-term objective of local governments was to raise the drainage base levels of the watercourses. This objective was fueled by a CBA performed by the national government, which claimed that raising drainage base levels would decrease the soil subsidence rates. Although profitable dairy farming would no longer be possible and large-scale transitions from dairy farming to nature restoration would be necessary, this disadvantage would be outweighed by a decrease of management costs (Van Brouwers-Haven and Lokker 2010). However, projects aimed at a top–down implementation of the transition in land use met with resistance from agricultural stakeholders. A lock-in situation developed, which made government organizations aware that more effective stakeholder collaboration was needed to produce legitimate results and develop viable management strategies. To aid this resolve, an assemblage of stakeholders (see appendix 3A) initiated a collaborative policy process to explore the effects of other strategies for water management and land use planning. To support this endeavor, they opted for a CBA as a heuristic aid, underpinned by quantitative modelling, and supplemented by a web-based discussion tool. In this paper, we use this case to demonstrate the added value of CBA in combination with a deliberative tool as a support system for collaborative policy processes.

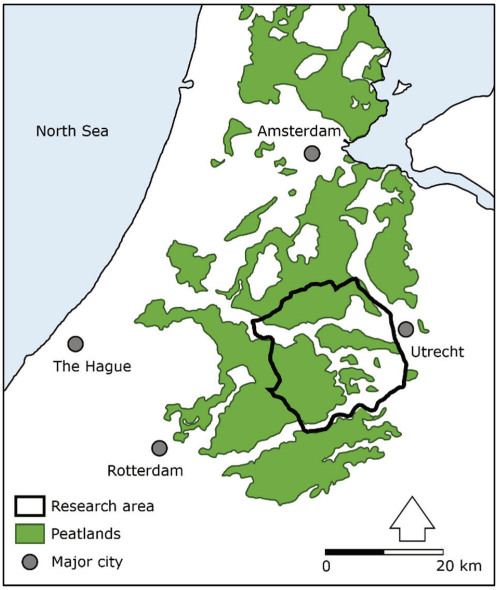

Fig. 3.1. Location of the research area in the western part of the Netherlands.

3.3 Methods

Outline of research

The research approach of the current study reflects the basic outline suggested by Dutch CBA guidelines. To some extent, the CBA practice in the Netherlands reflects a symbiosis of CBA and MCDA that allows for multiple evaluative endpoints. The reason for this is a comprehensive CBA guideline issued by the Dutch government at the turn of the century (Eijgenraam et al. 2000). Unlike traditional CBA, this guideline aimed not to present final cost–benefit ratios of project alternatives, but to give an overview of societal welfare effects. Some authors even remarked that "it functions more as a scorecard method for all relevant impacts than as a CBA" (De Jong and Geerlings 2003). In subsequent years, the guideline was proceeded with several refinements and additions (e.g. Faber and Mulder 2012; Romijn and Renes 2013). Although each guideline is slightly different, the basic approach consists of (a) a problem analysis, (b) the definition of project alternatives, (c) the assessment of costs and benefits, and (d) a clear presentation and interpretation of the results. All guidelines stress the importance of including an uncertainty analysis and a distribution of the effects among the affected stakeholder groups. In addition, some guidelines also stress the importance of including non-monetary effects and indirect effects, i.e., the wider economic effects for producers and consumers caused by the direct effects of a project alternative.

From the Dutch CBA guidelines, we derived a CBA approach of three consecutive phases, which we embedded in the deliberative decision-making process of the case study. The first phase reflects steps (a) and (b) of the CBA guidelines. We engaged researchers and practitioners to collaboratively define water management strategies and the timeframe of the assessments. The second phase reflects step (c) of the CBA guidelines. We used an integrated modeling framework to assess the physical effects of the management strategies. Subsequently, we assessed the ensuing costs and benefits, and distributed these effects to the stakeholder groups affected. We elaborated upon the suggestions of the guidelines by further redistributing the effects of the governmental stakeholders to all tax-payers in proportion to the taxes they pay. The third phase reflects step (d) of the CBA guidelines. We discussed all results with the advisory boards of the participating governmental organizations. We elaborated upon the suggestions of the guidelines by presenting trends in annual values in addition to Net Present Values (NPVs). Complementary to this, we used a deliberative web-based tool to evaluate the added value of our approach. In addition, we discussed options for follow-up strategies at several meetings attended by a broad range of stakeholders involved in peatland management. All meetings in the third phase focused on knowledge exchange between stakeholders and governments, culminating in a reconnaissance of shared interests.

Defining management strategies

In the first phase, the initiators of the policy process organized small-scale meetings to discuss peatland dynamics and plausible management alternatives with several organizations of researchers and stakeholders (see appendix 3A). In the final meeting, all participants jointly defined three water management strategies and the "business as usual" scenario (strategy 0) in which the current management is continued unchanged:

0. Current surface water levels. The drainage base levels of the watercourses are maintained at the same level relative to the ground surface, to facilitate the current land use. This means that the absolute surface water levels must be lowered periodically, to compensate for the soil subsidence. This management strategy reflects the present policy.
1. Progressively higher surface water levels. The drainage base levels of the watercourses are maintained at the same absolute level. This implies that as soil subsidence progresses, the drainage base levels rise relative to the ground surface. This management strategy was chosen because it reflects the former top–down approach to achieve a transition in land use.
2. Lower surface water levels. The drainage base levels in the watercourses are maintained at 90 cm below ground, to optimize conditions for agricultural land use. To achieve this, absolute surface water levels must be lowered periodically to compensate for the progressive soil subsidence. This management strategy was chosen because it facilitates current agricultural land use, regardless of future impacts on other interests, and is the opposite of strategy 1.
3. Current surface water levels, with field drains installed at depths below the surface water level. During wet periods the drains result in lower groundwater levels, because the water drains away faster. During dry periods, the drains supply water to the soil, resulting in higher groundwater tables. These drain-dependent groundwater tables have been reported to increase agricultural production and to retard soil subsidence (Querner et al. 2012). This management strategy was chosen as an alternative to higher drainage base levels.

We considered a timeframe from 2010 to 2100, including predicted climate change. Because predictions for climate change diverge substantially, we used an uncertainty range with a lower boundary at which no change occurs and an upper boundary according to the W^+ scenario of the Royal Netherlands Meteorological Institute, which is the most extreme regional projection of climate change for the Netherlands (Van den Hurk et al. 2006). Regarding demography and urbanization, we assumed the population and the extent of built-up areas would remain the same as in 2010.

Assessing the effects of management strategies

In the second phase, the physical effects of the water management scenarios and the ensuing costs and benefits were assessed (Fig. 3.2). Once again this required a joint effort, with researchers making the assessments and practitioners providing GIS input data and empirical knowledge. We used a spatially and temporally explicit modeling framework that simulates the interrelated dynamics of surface water levels, groundwater tables, and soil subsidence, as well as the ensuing effects on gardens, CO_2 emissions, the water system, real estate, agricultural land use, and recreational visitors (Van Hardeveld et al. 2017).

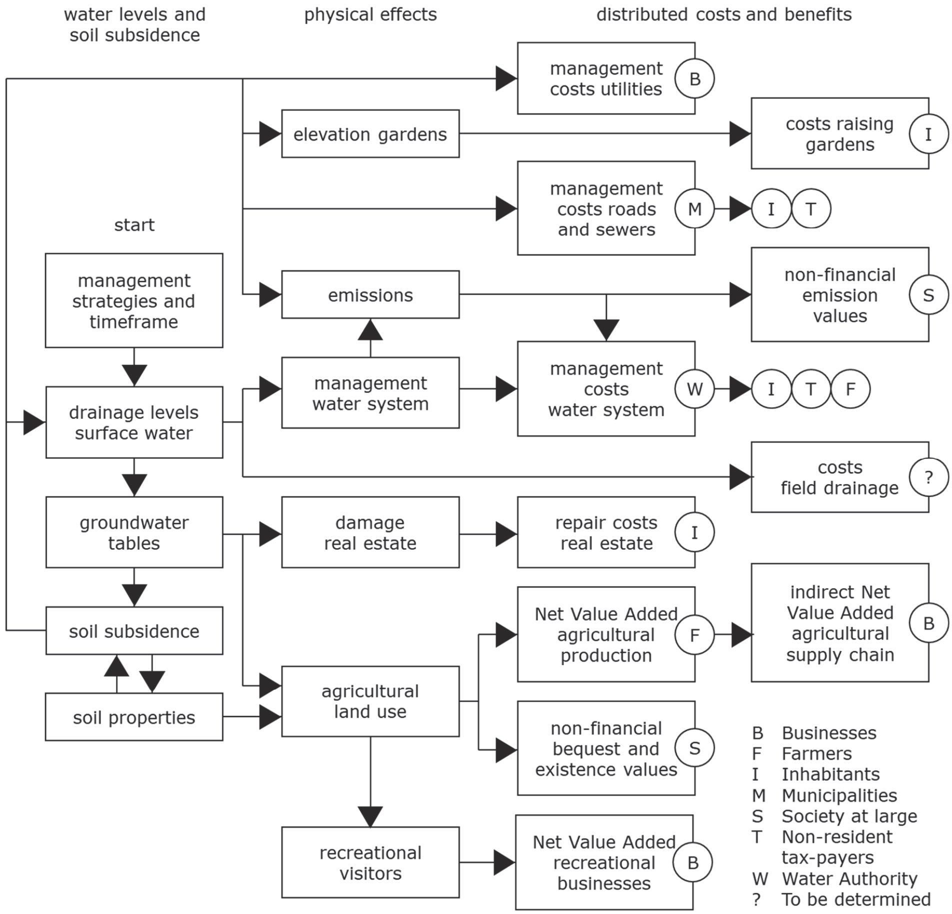

Fig. 3.2. Flow chart of the effects assessed in the current study. Arrows indicate the direction of the sequence. Circles indicate the stakeholders to whom the costs and benefits are distributed. The costs incurred by municipalities and the water authority are redistributed to the other stakeholders in proportion to their share of taxes.

The modeling framework assessed soil subsidence as a function of soil composition and groundwater tables, using an empirical equation adapted from Van den Akker et al. (2008). After each time step of five years, the amount of soil subsidence (in cm) was subtracted from the thickness of the top of the peat soil. Next, the drainage base levels in the watercourses were adjusted, to restore them to the same level relative to the ground surface. The altered elevation of the soil surface and the drainage base levels were used to calculate the change in groundwater tables.

The impacts on the emissions of CO_2 were assessed by relating the calculated soil subsidence to average peat soil properties in the research area (Van den Akker et al. 2008). Additionally, using empirical knowledge obtained from the regional water authority, the emission of CO_2 from the diesel-powered pumps that drain the watercourses was estimated. The calculated surface water levels combined with empirical knowledge of the regional water authority were used to assess the impacts on the management of the water system, i.e., the weirs, embankments, fish ladders, and pumping discharge required. Using the calculated groundwater tables combined with empirical data on foundations, the real estate damage was assessed by comparing the change in groundwater table since the year a house was built with the threshold for damage resulting from a change in the groundwater table assigned to the foundation type of the house. The calculated groundwater tables and soil properties were used to estimate the decline in crop and dairy yields. Above a certain threshold of yield loss, dairy farming was assumed to be replaced by biomass crops for energy production and ultimately by marshland. Subsequently, the number of recreational visitors was derived by combining the assessed land uses with empirical data on the number of recreational visitors for these land uses.

Using the results from the integrated modeling framework and empirical financial data, we determined the investment sums and maintenance costs required for (a) gardens, (b) utility cables, (c) roads and sewers, (d) the water system, (e) field drainage, (f) real estate, (g) the NVA of agricultural production, (h) the NVA of the agricultural supply chain, and (i) the NVA of recreational businesses. Assuming an interest rate of 4% we derived the annual management costs, which equal the sum of interest, depreciation, and maintenance. If the management costs calculated for the present situation did not correspond to actual government management budgets, we adjusted the empirical cost indicators to obtain a better fit. Moreover, if the empirical indicators provided converged, we used an uncertainty range, with the resulting mean annual management costs corresponding to actual government management budgets and a range of approximately 20% between the lower and upper boundaries. In addition, we assumed price levels would remain fixed at the average for the years 2009–2012. We were unable to obtain sound regional agricultural projections because long-term developments in global agricultural markets and the Common Agricultural Policy of the European Union are too uncertain, so instead of defining an arbitrary uncertainty range for the NVA, we assumed that market conditions would remain unchanged.

The effects on the agricultural supply chain are indirect effects of the management strategies, which, according to CBA literature, can only be included if they result in additional welfare effects, for instance by reducing costs due to market imperfections (SACTRA 1999; Eijgenraam et al. 2000; Vickerman 2007). However, it is very difficult to assess the degree to which market imperfections occur and additional welfare is generated (Rouwendal 2002; Hof et al. 2011). We therefore included the full scope of the indirect agricultural effects and assigned the task of delineation to the participants in the policy process. However, because it is not clear to what extent the indirect effects generate welfare, we presented them separately from the direct effects. In addition, indirect effects that clearly do not result in additional welfare effects were excluded. For instance, indirect benefits for government contractors equal the costs of government investments. Incorporating them would have meant they would have been valued twice, which is methodologically unsound and decreases the transparency of the CBA.

The non-financial values of the emission of CO_2 were derived by combining the calculated emissions with the rounded average price of CO_2 credits in the years 2009–2012. The non-financial bequest and existence values were derived with Willingness to Pay estimates. Using the guidelines for valid benefit transfer of Brouwers and Spanink (1999) and Bos (2007) we transferred the Willingness to Pay estimates obtained by a survey used for a similar policy process, i.e., an appraisal of a similar range of land use categories in the peatlands to the north of Amsterdam. In accordance with Bateman et al. (2006) we estimated the number of residents willing to pay as 47% of the residents within a radius of 10 km from the research area.

We derived the economic effects of the management strategies, i.e., the generated societal wellbeing, by comparing the assessed costs and benefits of the management strategies and the "business as usual scenario" (strategy 0). Many CBA guidelines recommend distributing the economic effects to assess the goal of social fairness (e.g. Eijgenraam et al. 2000; Romijn and Renes 2013). This is especially relevant if the preferred management strategy is not enforceable by the central government but instead requires a collaborative effort from multiple stakeholders. We therefore heeded the recommendations of the guidelines and distributed all economic effects to the stakeholder groups affected.

Because government management costs are funded from taxes, the economic effects for the regional water authority and the municipalities were redistributed to all tax-payers in proportion to the taxes they pay: farmers (4% of the water management costs), residents (33% of the water management costs and 76% of the management costs of roads and sewers), and tax-payers who do not reside in the peatlands but elsewhere in the area administered by the regional water authority or municipalities (63% of the water management costs and 24% of the management costs of roads and sewers). Because businesses do not pay taxes for the management costs considered, they were excluded from this redistribution. Note that farmers are residents too, but were considered a separate group because they have different stakes than the non-agricultural residents and pay extra tax to the regional water authority. The costs of field drainage were not assigned to any group, because these costs are usually paid for jointly by several groups of stakeholders, in varying ratios. Because all non-financial values are virtual values that do not result in monetary transactions, these values were assigned to society in general.

Traditionally, a CBA discounts all present and future economic effects at a positive constant rate, resulting in the NPV of all present and future values combined. Costs and benefits that do not occur within a few decades therefore seem inconsequential. However, collaborative policy processes entail multiple perspectives (in the sense of time horizons). Each collaborating stakeholder will value future developments from their own perspective. Whatever discount rate is chosen, most likely there will always be another discount rate that would have better reflected the perspective of some of the stakeholders. Therefore, we not only presented NPVs, but also trends in annual values, as this allows each collaborating stakeholder to interpret the information from their own perspective.

Discussing the results

In the third phase, we presented the balance of the effects of the management strategies, along with all constituent assessments. We discussed the results on four separate occasions with advisory boards of the initiators of the policy process, in the presence of approximately 100 people who represented governmental organizations and organized interest groups (see appendix 3A). Afterwards, we asked those who participated in the meeting to reflect on the results. We used a web-based tool to evaluate their agreement to statements regarding (a) the quality of the analyses, (b) the added value of the overall approach, (c) the use of annual values instead of NPVs, (d) the distribution of values, (e) the assessment of indirect economic effects for the agricultural supply chain, and (f) the added value of the economic valuation of non-financial effects. In addition, we inquired about the perceived need for a collaborative effort to implement adaptations in the follow-up phase, because this idea was frequently mentioned in the meetings. For comparison, we normalized their responses to a scale ranging from 0 to 1. We also included several open follow-up questions, to elicit the reasons for their opinions. The tool also allowed them to reflect on responses of other respondents and to discuss various opinions. From their responses we derived an overview of the most frequently mentioned reasons for positive and negative opinions about the statements.

We discussed the combined results of our assessments and the evaluation on eight separate occasions, which were attended by approximately 100 employees and board members of the participating organizations, and 40 representatives of other governmental organizations and organized interest groups involved in peatland management. At all meetings in the third phase, we deliberately did not draw specific conclusions from the CBA, but left the participants free to make their own tradeoffs between all criteria. Furthermore, we guided the discussions toward an exchange of standpoints and a reconnaissance of shared interests, instead of delivering verdicts on the management strategies.

3.4 Results

<u>Effects of the management strategies</u>

We found that the soil subsidence is highly dependent on the soil composition and the groundwater table, with limited subsidence at locations with high groundwater tables or sandy crevasse deposits but maximum subsidence exceeding 1 meter at locations with low groundwater tables and peat soil (Fig. 3.3). The differences in soil subsidence result in pronounced physical effects. For instance, the lower surface water levels make it necessary to construct 301–304 km of additional embankment, whereas progressively higher surface water levels result in no change at all, because the drainage base levels of the watercourses remain at the same absolute level. Another striking example is the area of dairy farming that is converted to biomass crops. Progressively higher surface water levels result in a conversion of 17–76 km², whereas lower surface water levels result in only 1–3 km². On both accounts the other water management strategies result in intermediate physical effects.

The economic effects over time differ regarding the management strategies and the affected stakeholders (Fig. 3.4 and Table 3.1; see appendix 3B for the non-distributed economic effects). Progressively higher water levels (strategy 1) result in a fall in revenues from dairy farming. Therefore, the economic effects for farmers (3.4D) and indirectly for businesses (3.4F) are negative. Simultaneously, the soil subsidence is diminished, which results in positive economic effects for all other stakeholders. For inhabitants (3.4B) and non-resident tax-payers (3.4C), this welcome outcome is mainly due to a reduction in management costs for the water system, the roads, and the sewers, which constitute the lion’s share of the economic effects for these stakeholder groups. Society at large (3.4A) also profits, because emissions decrease, and bequest and existence values increase. The lower water levels (strategy 2) have the opposite effect: revenues for dairy farming and soil subsidence both increase, whereas species abundance decreases. For both management strategies the direct effect for businesses (3.4E) is small because the average spending of recreational visitors in the area is relatively low and the predicted land use changes have a limited impact on the number of such visitors.

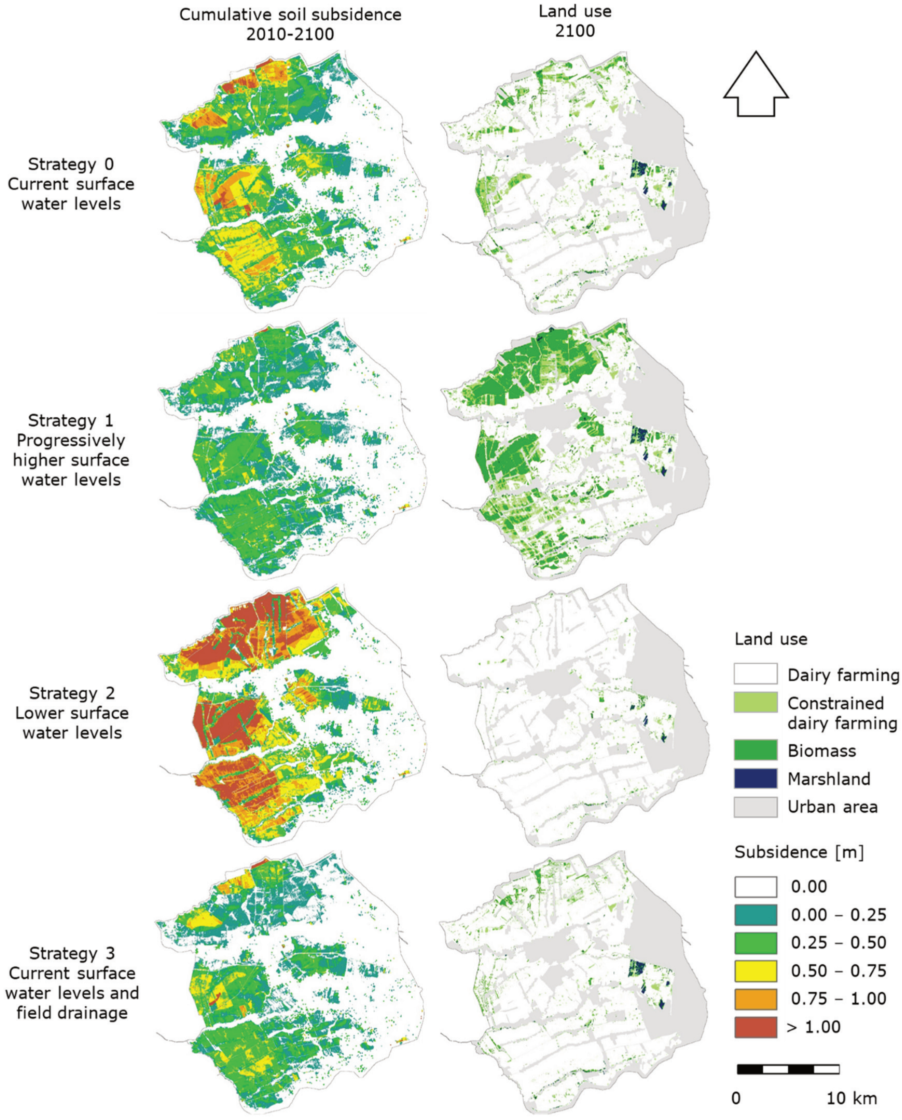

Fig. 3.3. Cumulative soil subsidence in 2010–2100 and agricultural land use in the research area in 2100. Note that soil subsidence only occurs at locations with peat soil. At locations where the groundwater tables have risen, the land use has changed successively from dairy farming to constrained dairy farming (i.e. more than 20% fall in crop yields), biomass crops, and, ultimately, marshland.

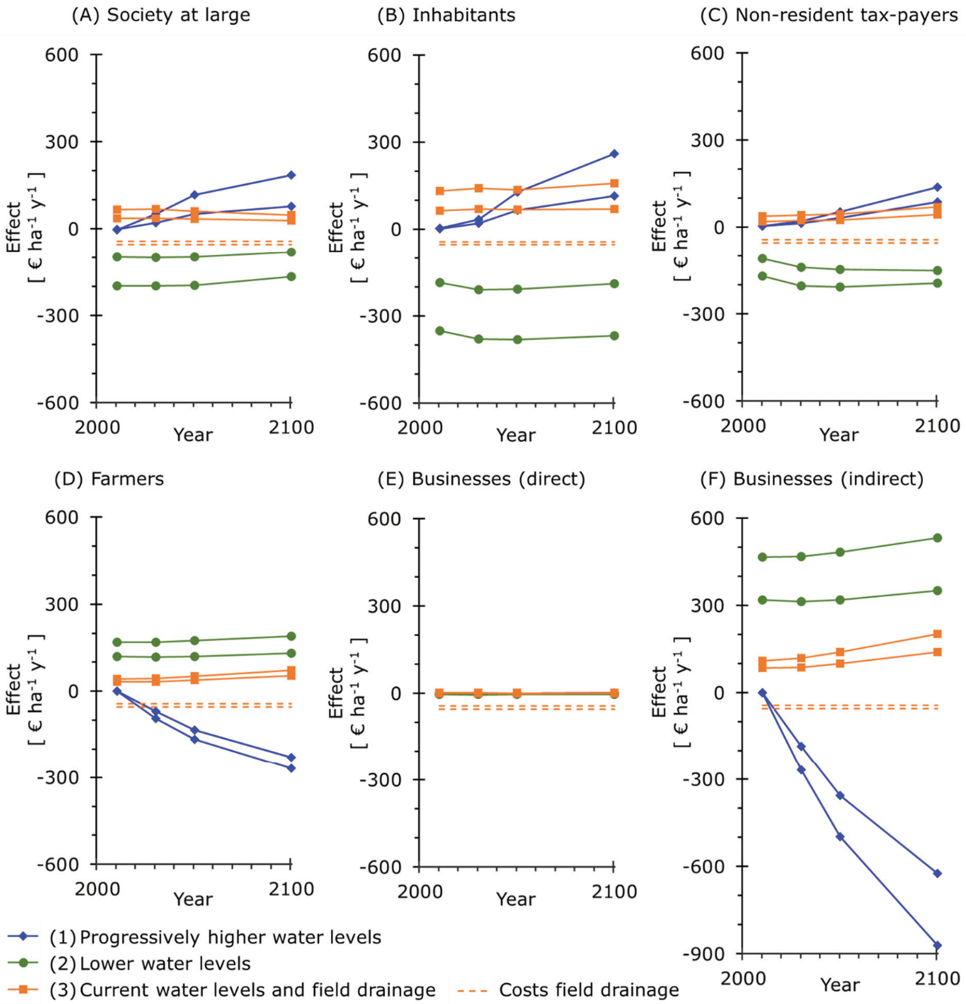

Fig. 3.4. Distributed economic effects of the management strategies, i.e., the differences compared with strategy 0. Both upper and lower limits to the range of values are shown. Note that the costs of field drainage are shown separately in all graphs, and the uncertainty regarding agricultural markets is not reflected in the results (D and F).

The effects of current water levels with field drainage (strategy 3) are more evenly distributed, with positive economic effects for all stakeholders. However, to implement this management strategy the negative economic effects of field drainage itself also need to be offset, which will diminish the overall economic effect of the affected stakeholders. It is noteworthy that although all stakeholders profit from this management strategy, only the positive indirect effects for businesses outweigh the negative effects of field drainage itself. Therefore, for most stakeholders it is not economically viable to implement this management strategy without the aid of other stakeholders.

Table 3.1. Distributed Net Present Values (million euro). Both upper and lower limits to the range of values are shown. Note that the uncertainty regarding agricultural markets is not reflected in the results. Non-monetary values that could not be assigned to a clear stakeholder group were left undistributed, i.e. assigned to "Society at large".

Management strategy	1	2	3
Society at large	11 — 28	-121 — -55	18 — 36
Inhabitants	17 — 33	-279 — -151	50 — 104
Non-resident tax-payers	12 — 18	-148 — -101	17 — 32
Farmers	-55 — -43	90 — 129	26 — 34
Businesses (direct)	0	-2 — -1	0 — 1
Field drainage	0	0	-41 — -33
Total	**-2 — 24**	**-421 — -217**	**78 — 166**
Businesses (indirect)	-114 — -162	240 — 357	68 — 91

Discussion of the results

23 attendees of the four meetings with the advisory boards (see appendix 3A) evaluated the quality of the analyses, the added value of the approach and its constituent elements, and the need for collaborative adaptations. Collectively, they issued 153 responses, which we grouped into 28 statements that elucidate their opinions (Table 3.2). Most viewed the CBA approach and its constituent elements favorably. The mean opinion score for the quality of the analyses was 69 out of 100. They frequently praised the approach's relevant and comprehensive assessments, i.e., the simultaneous assessment of multiple effects, all of which were underpinned by quantitative modeling. Criticism mostly concerned the scope of the scenarios and various minor flaws and omissions in the assessments. In addition, they frequently mentioned the difficulty of evaluating uncertain future developments and the effects of novel adaptations.

The mean score for their opinion of the added value of the overall approach was 77 out of 100. They especially appreciated the support given to the policy process. Most constituent elements of the approach were viewed favorably as well, with mean opinion scores ranging from 63 to 83 out of 100. The topic receiving the most positive response was the assessments of indirect economic effects, because this improved the comprehensiveness of the assessments. Because the main agricultural processor in the Dutch peatlands is a dairy cooperative to which many of the farmers belong, the farmers' interests are better captured by including the indirect benefits for agricultural processors in the CBA. The topic receiving the most negative response was the economic valuation of non-financial effects, with a mean opinion score of 25. The most frequently mentioned statement to elucidate the low opinion reflects the difficulty of interpreting these effects. Many practitioners suggested employing non-financial indicators and using multiple evaluative endpoints instead of a single economic endpoint. The mean opinion score for the need for collaborative adaptations was 73 out of 100. The need for a context-specific follow-up was stressed, but a complementary generic top–down approach was deemed necessary too.

We discussed the results of our assessments and the evaluation on eight separate occasions, which were attended by approximately 140 people involved in peatland management. These meetings guided the participating organizations in their design of peatland management strategies. Collectively, they decided that the former top–down approach to raise water levels and achieve a transition in land use (strategy 1) was not viable, because gains and losses were unequally distributed. Instead, the participating organizations decided to embark on several context-specific follow-up processes primarily aimed at applications of field drainage (strategy 3), in which the management costs were shared collectively by the participating stakeholder groups.

Table 3.2. Overview of the participants' opinions of the research, and the statements they gave to elucidate their opinions. Opinions are normalized to scores ranging from 0 (low) to 100 (high). Beside the mean score, the standard deviation is given in parentheses. The number in parentheses after each statement indicates how many participants issued that statement.

Topic	Opinion score	Participants' statements to elucidate low opinions	Participants' statements to elucidate high opinions
Quality of the analyses	69 (±19)	• The assessments need improvements (8) • The scenarios need a broader scope (8) • The uncertainty of long-term developments is difficult to interpret (7) • More knowledge is needed on the effects of adaptations (7) • A comparison is needed with other locations (3) • The transparency needs improvements (3)	• The great comprehensiveness of the assessments is good (7) • The transparency is good (2)
Added value of the overall approach	77 (±29)	• Not all elements are equally important (3)	• The process is supported by stakeholders (5) • The discussion of standpoints is stimulated (2)
Added value of trends in annual values	63 (±35)	• Net Present Values are also useful (3)	• Improved understanding of long-term developments (6)
Added value of distributed values	76 (±31)	• Questions the fairness of the taxes levied (8) • Distracts from overall societal wellbeing (1)	• Is relevant for a discussion on tradeoffs (9)
Added value of indirect economic effects	83 (±24)	• The assessment of indirect effects needs a broader scope (7) • The assumptions of indirect effects are uncertain (7)	• The comprehensiveness of the assessment is improved (5)
Added value of the economic value of non-financial effects	25 (±30)	• The interpretation is difficult (10) • Non-financial effects are important (9) • Separate indicators are needed for non-financial effects (7)	• Economic values are easy to compare (4)
Need for collaborative adaptations	73 (±22)	• A generic top–down approach is also needed (6) • Conservative stances are problematic (2) • The setting must feel safe for collaboration (1)	• A context-specific follow-up is needed (9) • A collaborative approach is needed (4)

3.5 Discussion

Added value of the approach

As stated in the introductory section, several scholars believe the combination of CBA and deliberative tools has high potential to support collaborative policy processes (De Jong and Geerlings 2003; Turner 2007; Brown and Ryan 2011; Beria et al. 2012). However, the effectiveness of these combinations depends on their ability to overcome process-related issues associated with CBA: such as misunderstandings and inadequate communication between CBA practitioners. In this section, we discuss to what extent these process-related issues were dealt with in the case study, and which limitations of the approach we perceive. In addition, we will present some limitations of our approach and suggestions for further research.

In the case study, the CBA results were deliberated at length. The participants predominantly regarded the research and its constituent elements favorably (Table 3.2), which we believe is a clear indication of a successful CBA process. A further indication that process-related issues were handled adequately is the nature of the discussions. Although many statements were issued about methodological limitations (Table 3.2), these issues did not dominate the discussions or kindle a sense of mistrust among the participants. Instead, the discussions focused on the management problem at hand, propagating the need for collaborative adaptations and context-specific follow-up processes.

Beside the abundant opportunities to deliberate results, we perceive two main reasons for the successfulness of the CBA process: (1) the design of the process and (2) the presentation of the results. The assembled organizations that initiated the process were very clear from the start that they intended to use the CBA as a heuristic aid. Such explicit communication about the envisioned role of CBA in a decision-making process is believed to contribute to the prevention of process-related issues (Mouter et al. 2013). In addition, a participatory approach for designing management strategies was used. Also, empirical economic data from the stakeholders were used for the assessments. We believe these design characteristics of the CBA process increased the support of the participants, which was the most frequently stated reason for the added value of the overall approach (Table 3.2).

Regarding the presentation of results, we aimed to increase the transparency of the assessments by unraveling the composite CBA result, i.e., the cumulative NPVs of the management strategies. Beside the cumulative NPVs, we also presented the results from the quantitative modeling framework that we used to underpin the CBA (Fig. 3.3), trends of annual values in time (Fig. 3.4), and distributed values (Fig. 3.4 and Table 3.1). Consequently, our approach presents costs and benefits as multiple evaluative criteria. For instance, beside the cost–benefit ratio of direct effects, our approach also considers a cost–benefit ratio for indirect effects. Furthermore, the emphasis on distributed values and trends in annual values allows the user to weigh the constituent components of the cost–benefit ratios. Arguably, this is key to ensure strong support for collaborative policy processes. The Kaldor–Hicks compensation principle implies that from the perspective of overall societal wellbeing it suffices to draw conclusions from the cumulative NPVs of management strategies, regardless of whether the gainers compensate the losers. But if the preferred management strategy requires a collaborative effort from multiple stakeholders, we argue that a prerequisite for a successful policy process is a discussion on equitable tradeoffs. We therefore allowed the participants to exchange standpoints on how they value the future developments, which opportunities they perceive for improving wellbeing by means of indirect economic effects, and which distribution of costs and benefits they regard as fair.

<u>Limitations of the approach and suggestions for further research</u>

Even though the overall approach was judged favorably by the participants of the case study, several limitations emerged as well. The most notable limitation was the assessment of the economic value of non-financial effects. The participants clearly stated that these effects were difficult to interpret (Table 3.2). Their opinions reflect previous critiques that such valuations narrow down non-financial values to current individual preferences and instrumental utility maximization goals (e.g. Costanza 2006). Therefore, CBA and/or MCDA should have a broader, more holistic scope, including less tangible and indirect impacts (Browne and Ryan 2011; Beria et al. 2012). We endorse this suggestion, because the participants in our case study clearly expressed a preference for separate indicators for non-financial effects (Table 3.2). In view of these results, we suggest that further research should aim to capitalize upon these revealed preferences.

A further research suggestion is the enhancement of the collaborative interpretation of the results. In our case study, we witnessed how the participants increased their capacity to understand the complexities of the peatlands and their ability to explore its potentialities by exchanging knowledge among themselves and with researchers. However, the participants often found it difficult to interpret the uncertainty caused by various assumptions and future developments (Table 3.2). This predicament can easily result in methodological debates that distract from the management problem at hand, creating the process-related issues that have been reported in previous CBAs (Beukers et al. 2012; Mouters et al. 2013). To avoid this pitfall, we believe it necessary to further increase the transparency of the approach, as well as its ability to handle uncertainty in information-rich context-specific processes similar to those in our case study. In this regard, we specifically draw attention to virtual learning platforms and geo-technology, which are designed to analyze large amounts of data, and have been shown to enhance knowledge co-production and learning (Medema et al. 2014; Pelzer et al. 2016).

The question remains what the added value of our approach would be in other settings. In the case study, most participants were cooperative throughout the entire process, because they all had recently experienced the drawbacks of a non-participatory top-down approach to water management and land use planning in Dutch peatlands. However, a few propagandists of the former top-down strategy used the results strategically, by discrediting the transparency of assessments which results did not match their agenda and refusing any further discussion of the subject. Strategic use of knowledge like this by non-cooperative stakeholders is a commonly reported problem in science–policy interactions (Van Enst et al. 2014). If more participants would behave non-cooperatively like this, a deadlock would occur where each participant advances their own arguments without listening to those of others, a so-called “dialogue of the deaf” (Van Eeten 1999). Arguably, it is unlikely that our approach would be effective in such settings, where non-cooperative attitudes of participants persist regardless of opportunities to exchange opinions with other participants. In those settings, we would suggest a combination of CBA (or other analytical tools) with tools aimed at mediation instead of deliberation (Driessen and Vermeulen 1995).

3.6 Conclusion

To contribute to our collective understanding which tools can support the management of social–ecological systems in diverse situations, we demonstrated how quantitative modeling, CBA, and a web-based discussion tool were employed to support a collaborative policy process in Dutch peatlands. We did not use CBA as a decision-rule, to determine the optimal cost–benefit ratio of project alternatives, but as a heuristic aid, aiming for an intersubjective assessment of the preferred project alternative.

The stakeholders participating in our case study appreciated the approach's relevant and comprehensive assessments, and the ensuing multi-criteria discussion of costs and benefits. In the case study, we witnessed how our approach increased the capacity of the participants to understand the complexities of the peatlands and their ability to explore its potentialities. The analytical merits of CBA, underpinned by quantitative modelling, exposed that the former top–down approach to raise water levels and achieve a transition in land use was not viable, because gains and losses were unequally distributed. Although the participants perceived many methodological limitations, these issues did not dominate the discussions or kindle a sense of mistrust among them. Instead, their discussions focused on the management problem at hand, propagating the need for collaborative adaptations and context-specific follow-up processes. We believe this result can be attributed to (a) the clear, participatory design of the CBA process, (b) a comprehensive presentation of the constituent elements of the CBA result, and (c) the abundant opportunities to deliberate the results.

Although our case study demonstrates that the combination of CBA with a deliberative tool can support the reconnoitering phase of a policy process with cooperative participants, we do not propagate our approach as a one-size-fits-all solution for the support of the management of social–ecological systems. Arguably, it is unlikely that our approach would be effective in settings with non-cooperative stakeholders. Therefore, we call on other researches to share empirical insights that demonstrate which tools can support the management of social–ecological systems in those situations.

Acknowledgments

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Appendix 3A. Background of the participants in the policy process

The policy process was initiated by an assemblage of a regional water authority, two provinces, and a steering committee for the peatlands. They received input from two scientific research institutes, a consultancy company, and five municipalities. Table 3A.1 provides some background on these organizations, including their role in the policy process. Table 3A.2 provides an overview of the backgrounds of the participants that evaluated the approach.

Table 3A.1. Participants in the case study.

Organization	Background	Role in policy process
Copernicus Institute of Sustainable Development	Utrecht University institute for sustainability research and teaching	Researcher (advisor approach)
LEI	Wageningen University institute for agricultural research	Researcher (advisor on assessment of physical and economic effects; supplier of input data)
Grontmij	Consultancy company	Researcher (advisor on assessment of soil subsidence)
Water authority "Hoogheemraadschap De Stichtse Rijnlanden"	Regional government organization for water management in the research area	Practitioner (initiator of process; supplier of input data)
Provinces of Utrecht and South Holland	Regional government organizations for spatial planning and environmental quality in part of the research area	Practitioner (initiator of process)
Regional Committee "Stuurgroep Groene Hart"	Boundary organization for government organizations and societal stakeholders in the peatlands, aimed at supporting initiatives that strengthen the vitality and sustainability of the peatlands	Practitioner (initiator of process)
Municipalities of Woerden, Bodegraven–Reeuwijk, Gouda, De Bilt, and Wijk bij Duurstede	Local government organizations within or near the research area	Practitioner (supplier of input data)

Table 3A.2. Background of the participants that evaluated the approach.

Background	Organization
Water authorities in peatland areas	• Hoogheemraadschap De Stichtse Rijnlanden • Hoogheemraadschap van Delfland • Hoogheemraadschap van Rijnland • Hoogheemraadschap van Schieland en de Krimpenerwaard • Rijkswaterstaat • Waterschap Vallei en Veluwe • Wetterskip Fryslân
Other government organizations in peatland areas	• Province of Friesland • Province of North Holland • Province of South Holland • Municipality of Woerden • Environmental agency "Milieudienst Rijnmond"
Organized interest groups	• Village Committee of Oud-Kamerik • Dutch Federation of Agriculture and Horticulture • Interest group for municipalities in peatlands "Platform Slappe Bodem" • Interest group for peatland residents "Stichting Groene Hart" • Ecological interest group "Initiatiefgroep natuurbeheer Delft" • Cultural heritage interest group "Bond Heemschut" • Peatland Innovation Center • Freelance advisor on landscape quality

Appendix 3B. Additional results

Table 3B.1. Net Present Values of the effects of the management strategies (million euro). Both upper and lower limits to the range of values are shown. Note that the uncertainty regarding agricultural markets is not reflected in the results.

Management strategy	1	2	3
Water system	13 — 16	-124 — -103	6 — 8
Roads and sewers	13 — 28	-262 — -127	51 — 107
Real estate	1 — 2	-6 — -3	2 — 6
Gardens	2 — 4	-35 — -18	8 — 15
Agricultural production	-55 — -43	90 — 129	26 — 34
Utilities	0	-3 — -1	0 — 1
Recreational businesses	0	0	0
Emission values	5 — 10	-80 — -40	18 — 37
Bequest and existence values	7 — 19	-40 — -15	-1 — 0
Field drainage	0	0	-41 — -33
Total	**-2 — 24**	**-421 — -217**	**78 — 166**
Agricultural supply chain	-114 — -162	240 — 357	68 — 91

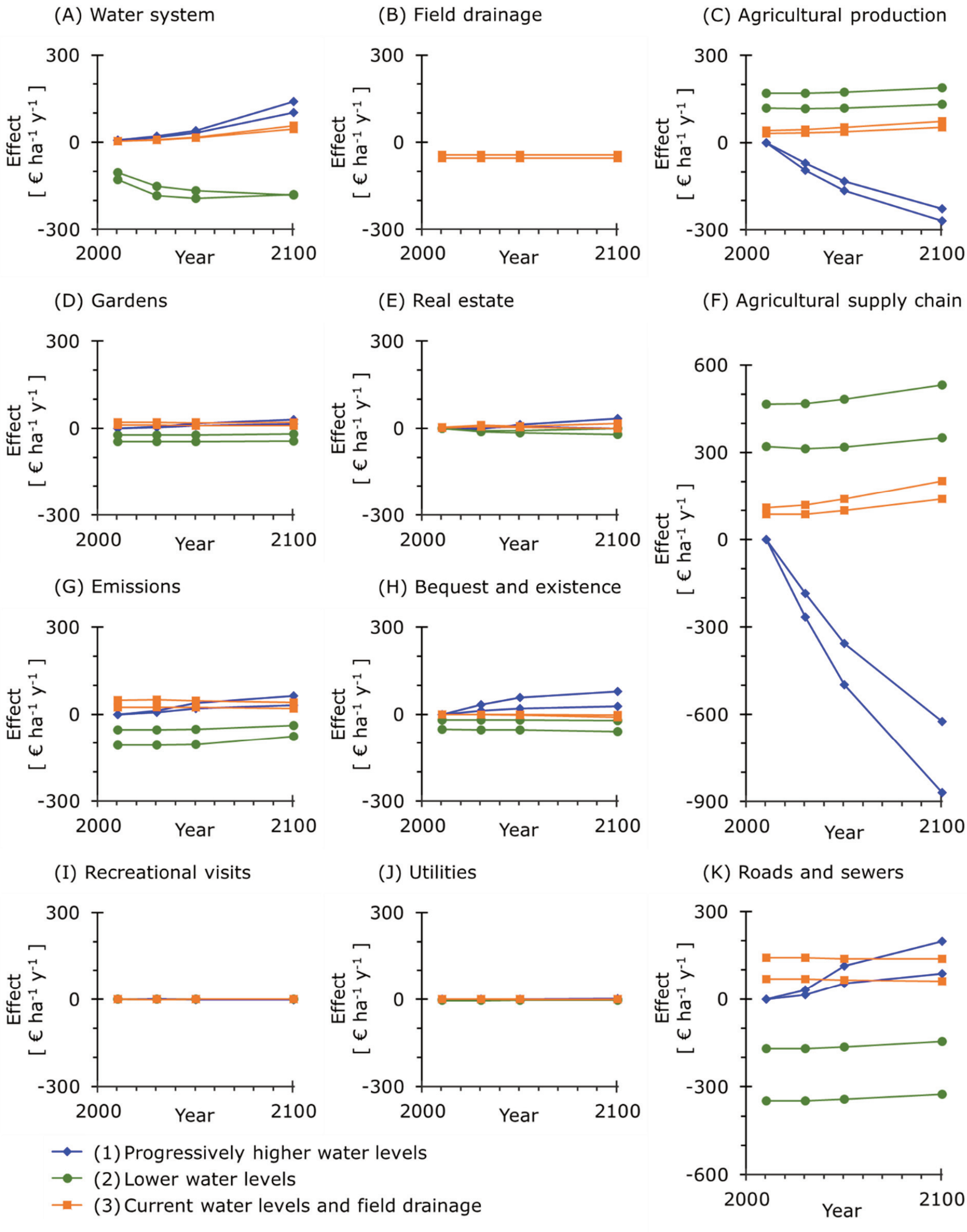

Fig. 3B.1. Economic effects of the management strategies, i.e., the differences compared with strategy 0. Both upper and lower limits to the range of values are shown. Note that the uncertainty regarding agricultural markets is not reflected in the results (C and F).

Chapter 4
How valuing cultural ecosystem services can advance participatory resource management: the case of the Dutch peatlands

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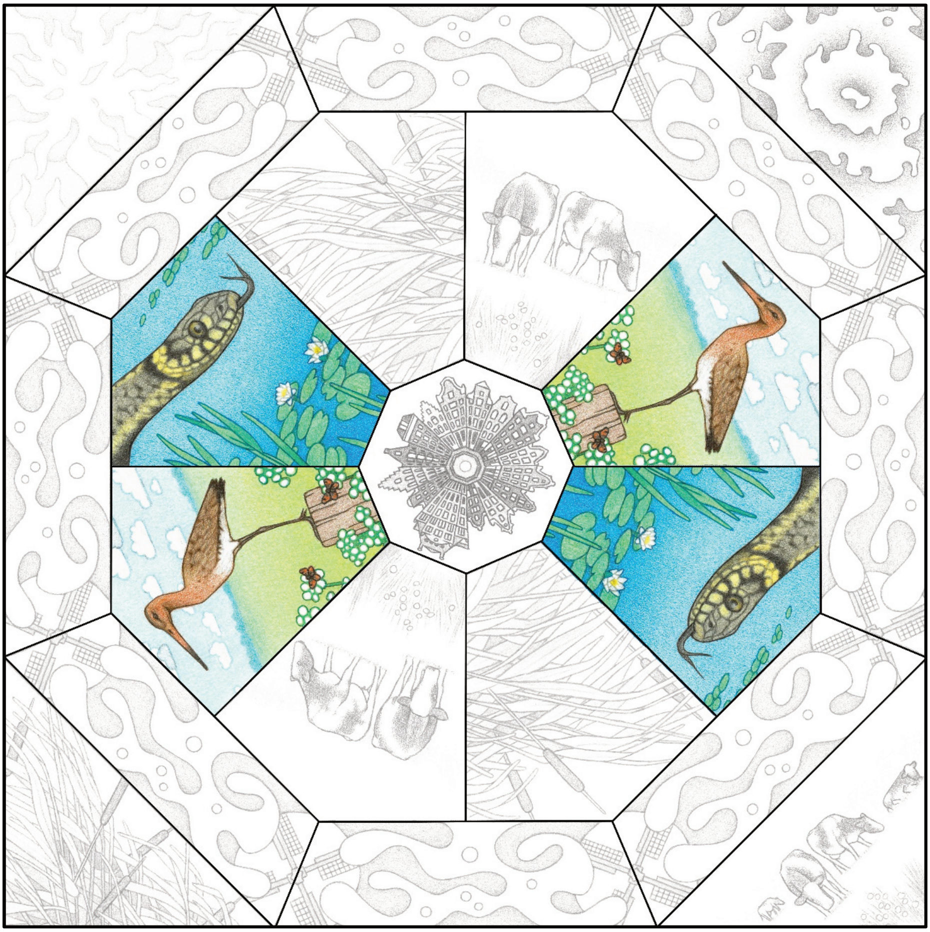

Currently, the Dutch peatlands are a major European stronghold for meadow birds. In the figure, this is depicted by the polygons with the black-tailed godwit. To slow down soil subsidence, water levels must be raised. As a result, some meadows might have to be converted to swamps, which will affect the ecology and the aesthetic values of the Dutch peatlands. In the figure, this is depicted by the polygons with the grass snake.

Abstract

To enhance the contribution of ecosystem services assessment to resource management, scholars advocated transdisciplinary approaches. This paper aims to contribute to our understanding of which approaches are best suited for different contexts and scales, by demonstrating how Cost–Benefit Analysis and a participatory non-monetary valuation of cultural ecosystem services were employed in the Dutch peatlands. In participatory workshops, we designed management scenarios and selected indicators for cultural ecosystem services. We successfully engaged the key stakeholder groups of the research area and surveyed their willingness to pay for the scenarios and their non-monetary valuation of the cultural ecosystem service indicators. We used the survey results to construct maps of the cultural ecosystem service valuation by the stakeholder groups, revealing the spatial consequences of their preferences that were not explicit beforehand. In addition, we analyzed the costs and benefits of the scenarios. The combined results supported a deliberation across multiple perspectives and value dimensions. The stakeholders appreciated the approach's ability to collaboratively explore pathways for putting sustainable management scenarios into practice. We discuss the added value of our approach, and point out how an integration with deliberative valuation techniques may further enhance its support to local planning processes.

4.1 Introduction

The concept of "ecosystem services" (ESs) has become increasingly influential in transforming environmental science and policy around the world (Chaudhary et al. 2015). However, several knowledge gaps remain, especially regarding more effective contributions to resource management and decision-making (Guerry et al. 2015). To advance the impact of ES assessments, many authors stress the need for integrated decision-making approaches that consider both monetary and non-monetary valuations, encompassing intersubjective and multidimensional cultural values (Beria et al. 2012; Daniel et al. 2012; Guerry et al. 2015; van den Belt and Stevens 2016; Jacobs et al. 2016). In this paradigm, the main aim of valuation shifts from underpinning trade-off analysis toward developing a shared understanding and dialog about plural values (Kenter 2016a).

A common approach for integrating monetary and non-monetary valuations is the use of multiple evaluative endpoints, drawing on the traditions of multi–criteria analysis and Cost–Benefit Analysis (CBA). For example, in their impact assessment of agricultural land-use changes on ESs in the UK, Bateman et al. (2013) used a non-monetary biodiversity index to complement monetary valuations of agricultural production, the greenhouse gas balance, recreational visits, and urban green space amenity. And Sijtsma et al. (2013) proposed to complement economic project appraisals with non-monetary assessments of the ESs that impact ecological quality and human health. Although the number of case studies with similar approaches is rapidly expanding, still relatively few examples exist of integrated approaches that incorporate assessments of cultural ESs (CESs), i.e. the non-material benefits that arise from human–ecosystem relationships.

Except for the easily marketable service of recreation and ecotourism, the valuation of CESs is considered particularly challenging because of their intangible and incommensurable properties (Chan et al. 2012; Daniel et al. 2012). The incorporation of CESs in integrated decision-making approaches requires valid and intuitively accessible non-monetary CES indicators to which values can be assigned. There is a broad science base for operationally defining CES indicators, but many CES indicators are deficient, for instance because spatially explicit information is lacking, or local stakeholders are not involved in their conceptualization (Hernández-Morcillo et al. 2013). To improve indicator quality, participatory mapping tools can be used to elicit place-based information and reveal to what extent values are shared by various stakeholder groups. For example, Plieninger et al. (2013) engaged the villagers of a cultural landscape in East Germany to map CESs and disservices, and Darvill and Lindo (2015) used participatory GIS to map provisioning and cultural ES indicators in a rural Canadian catchment across seven stakeholder groups. Newton et al. (2012) used participatory mapping to elicit "hotspots" of CESs across a catchment in the UK and demonstrated how to use this information to evaluate restoration strategies in conjunction with monetary assessments of provisional ESs and a non-monetary assessment of biodiversity.

Another challenge for CES valuation approaches is their ability to support deliberation between stakeholders. It is increasingly recognized that CESs reflect shared values that transcend individual preferences and do not exist a priori but are formed through deliberative processes (Irvine et al. 2016). Consequently, aggregated individual values may significantly differ from group-deliberated values (Kenter 2016b). CES valuation approaches need to acknowledge this conception of shared values and should allow for value deliberation between stakeholders.

Although there is a growing body of literature on CES assessment, many authors still call for further examples that advance our understanding of which valuation approaches are best suited to support participatory decision-making in different contexts and at different scales (Daniel et al. 2012; Van den Belt and Stevens 2016; Jacobs et al. 2016). Therefore, this paper aims to contribute to our collective understanding by demonstrating how participatory non-monetary CES valuation and CBA were employed to support a local resource management process in a part of the Dutch peatlands. The question we seek to answer is how CES valuation can be used to advance participatory resource management at a local scale. In addition, we will discuss the strengths and weaknesses of our approach and point out avenues to effectively engage stakeholders.

4.2 Background: the research area

The case we used for our endeavor is a participatory planning process of land use and water management scenarios in a part of the Dutch peatlands. We focused on an area of 24 km² in the peatland area in the west of the Netherlands, where the predominant land use is dairy farming (Fig. 4.1). During the Middle Ages, the natural fens in the area were drained and converted to arable fields and meadows by constructing polders: artificial catchments with a dense network of watercourses. The surface water levels in the watercourses determine the depth of the water table below ground, which steers the soil subsidence rate. The cumulative soil subsidence since the Middle Ages amounts to approximately 2 m (Schothorst 1977), resulting in the current land elevation ranging from 0.5 m above to 2.0 m below sea level. Although the peatlands have been subsiding for centuries, their medieval water system and allotment patterns still exist: long, narrow plots bordered by ditches, alternating with two small villages and three linear settlement zones of farmsteads and houses. The total population in the area is 4,200.

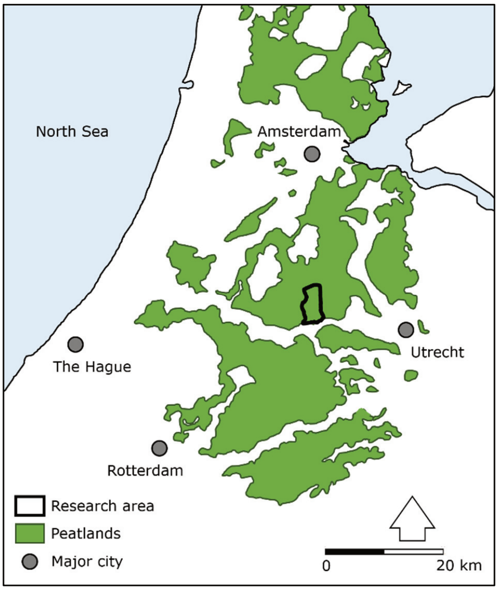

Fig. 4.1. Location of the research area in the western part of the Netherlands.

The overriding problem in the area is soil subsidence, which affects the regulating ES "erosion regulation". The soil subsidence results in high management costs for roads, sewers, utility cables, gardens, and the water system. It also results in real estate damage, and affects the emission of greenhouse gasses, thereby affecting the regulating ES "climate regulation". Although the undesirable effects can be diminished by raising the water levels, thereby reducing the rate of soil subsidence, this will reduce crop yields and hence affect the provisioning ES "food" (Van Brouwers-Haven and Lokker 2010; Van Hardeveld et al. 2017).

The long-term strategy for land use and water management in this area will have to address a broad and diverse range of values for all stakeholders. It should include not only the monetary costs and benefits of soil subsidence, but also the non-monetary CESs of the area. Changes in land use and water management may impact the medieval allotment patterns, which are acclaimed as valuable Dutch cultural heritage (Curtis and Campopiano 2014). Moreover, land use and water management also affect the breeding and chick rearing of meadow birds, for whom the Dutch peatlands are a major European stronghold (van der Vliet et al. 2014). Furthermore, an estimated 150,000 people visit the research area each year for low-intensive recreation activities such as bird watching, hiking, and cycling.

4.3 Methods

Outline of the research approach

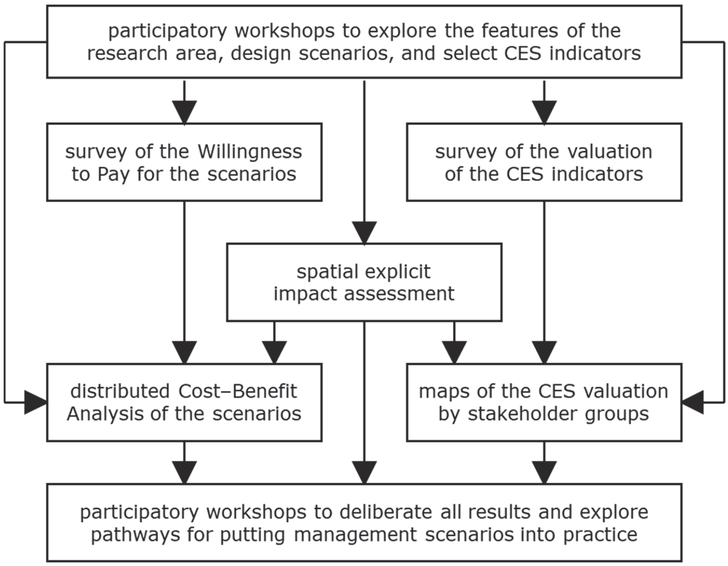

Fig. 4.2. Flow chart of the research approach of the current study. Arrows indicate the direction in which the research components feed into each other. CES = Cultural Ecosystem Service.

Our research approach consisted of five sequential steps (Fig. 4.2). First, we organized a sequence of three participatory workshops with local experts in which we explored the features of the research area, and designed scenarios for land use and water management with diverging impacts on the research area. To map the impacts on the CESs, we selected robust, transparent, and stakeholder-relevant CES indicators. Second, we surveyed the stakeholders' willingness to pay (WTP) for the current situation and the scenarios, as well as their valuation of the selected CES indicators. Third, we assessed the spatially explicit physical impacts of the current situation and the scenarios. Fourth, we used the designed scenarios, the results from the survey and the impact assessment to assess the resulting costs and benefits of the scenarios for several stakeholder groups, and construct maps of the CES valuation by the stakeholder groups. Fifth, in workshops with stakeholders who participated in the survey we used the DialogueMaps tool to deliberate all the results and explore pathways for putting sustainable management scenarios into practice.

Designing scenarios and selecting indicators

The first step of our research approach consisted of a sequence of three participatory workshops. In the first workshop (December 8th, 2014), we discussed the scope of the research with three peatland experts of the local municipality, the National Cultural Heritage Agency, and an interest group for cultural heritage. The local municipality was primarily concerned about their management options in response to agricultural developments, which on the one hand might reflect a scale enlargement of dairy farming, and on the other hand a conversion in land use to adapt to higher water levels. These developments would have divergent impacts on the ESs, the management costs of the municipality and the water authority, and the distribution of costs and benefits over stakeholder groups. The key stakeholder groups the local municipality wanted to engage in a dialog about these developments were (a) the farmers, i.e., the main land owners, (b) the inhabitants of the villages and settlement zones, and (c) the recreational visitors.

To accomplish this, we opted for a multi-phase participatory process. The chosen goal of the first phase was to initiate a process of broadly informing stakeholders about CESs and developing a dialog on planning options from the perspective of three key stakeholder groups. We envisioned the first reconnoitering phase would be followed by subsequent phases in which assessments would be more detailed. For the first phase, we decided to use a deliberative tool called DialogueMaps, which draws from experiences with a mediation and negotiation tool for the early phases of participatory land use planning processes (Janssen et al. 2005), using stakeholder-specific valuations of landscape-features to assess patterns of CES provision. To prepare for this endeavor, we organized two additional workshops.

In the second workshop (March 2nd, 2015) we asked thirteen local experts on ecology and cultural heritage from governmental organizations and interest groups to reflect on the results of the first workshop. They concurred with our selection of key developments, and suggested to use a timeframe of 50 years, which on the one hand allows for the gradual impacts of soil subsidence to manifest themselves, and on the other hand is still relevant for the long-term policy of the local municipality. Next, we discussed which features of the research area were most important to include in the research. This resulted in a comprehensive list of features that relate to the CESs in the research area. In addition, they suggested to value "modern livestock barns" and "agricultural traffic" as well, because they believed these features to have a big negative impact on the CESs in the research area.

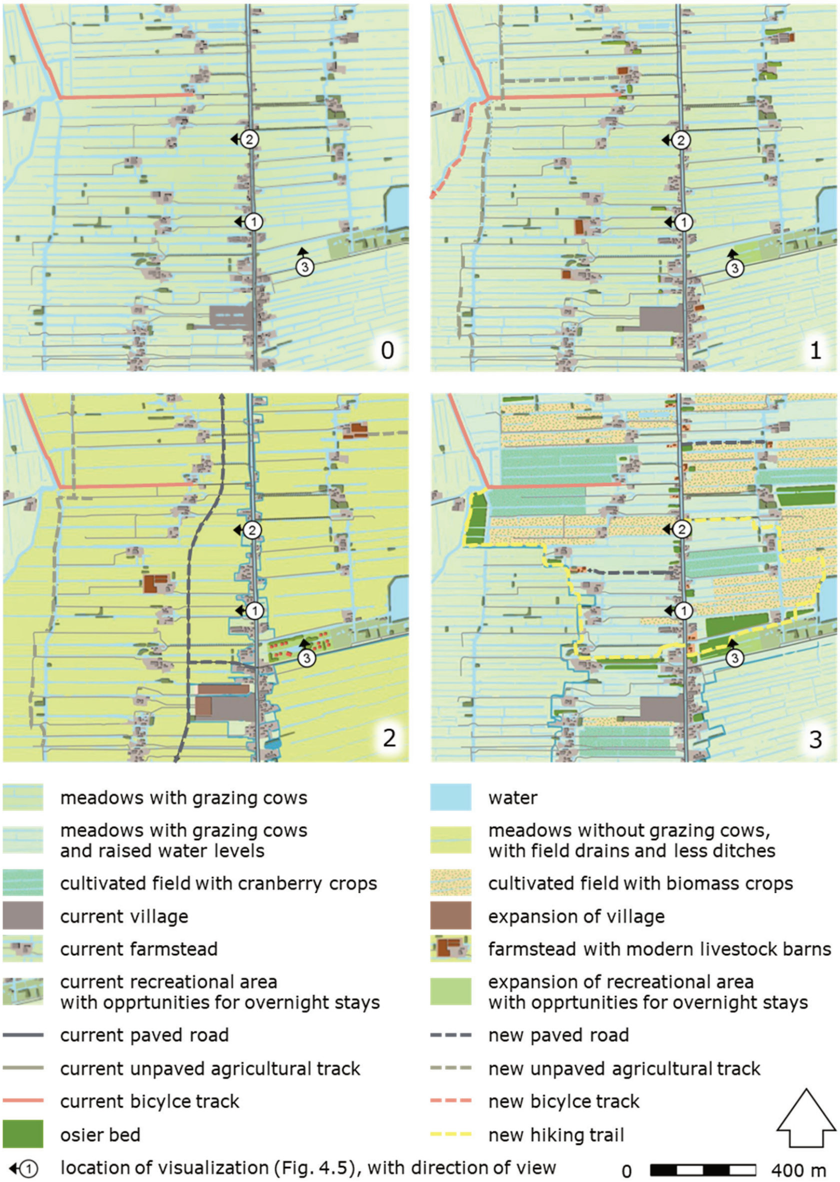

Fig 4.3. Topographic maps of the central part of the research area for the current situation (0), the "Business as usual" scenario (1), the "Scale enlargement of current land uses" scenario (2), and the "Wetter conditions" scenario (3).

Next, we selected CES indicators with which the impacts of the scenarios on CESs could be mapped. We deliberately limited the number of CES indicators, allowing for a valuation survey that respondents could complete within 10 minutes. Because the mapping of CESs is pivotal for our research approach, the CES indicators had to be robust, transparent, and stakeholder-relevant (Willemen et al. 2015). To meet these requirements, we selected four CESs of the Millennium Ecosystem Assessment (2005) classification system for ESs that related to features of the research area that were frequently highlighted by the participants of the second workshop, and were mappable and quantifiable using stakeholder-based valuations at the local scale of our research area: "sense of place", "cultural heritage values", "aesthetic values", and "recreation and ecotourism". We selected three indicators for each CES, which could be mapped spatially accurately, could be easily explained in layperson's terms, and could be quantified using stakeholder-based valuations (Table 4.1). Following up on the suggestions of the participants of the second workshop, we also selected two important negative impacts on the CESs—"number of modern livestock barns" and "length of roads with much agricultural traffic"—because this could support the design of management scenarios that minimize these impacts.

Table 4.1. Selection of cultural ecosystem service indicators and negative impacts.

Cultural ecosystem service	Indicators and negative impacts	Reason for selection
Sense of place	• Area of meadows	• Main land use
	• Area of cultivated fields	• Important accompanying land use
	• Area of meadows with grazing cows	• Highly visible, appealing species
Cultural heritage values	• Number of ditches which are part of medieval ditch patterns	• Key feature of historical geography
	• Number of old farmsteads	• Key feature of built heritage
	• Area which likely contains archeological remains	• Key feature of archeology
	• Number of modern livestock barns	• Big negative impact on the ensembled cultural heritage values
Aesthetic values	• Area visible from the main road and the main hiking trail	• Key feature of the aesthetic value of Dutch peatlands
	• Area of osier beds	• Important accompanying, alternating feature of openness
	• Area with high ecological quality	• Key feature of the aesthetic value of Dutch peatlands
Recreation and ecotourism	• Length of hiking trails	• Key reason for recreational visits
	• Length of cycling tracks	• Key reason for recreational visits
	• Area with opportunities for recreational overnight stays	• Important activity accompanying hiking and cycling
	• Length of roads with much agricultural traffic	• Big negative impact on opportunities for hiking and cycling

Surveying stakeholders' valuation

In the second step of our research approach, we surveyed the stakeholders' valuation of the CESs and their WTP for the current situation and the scenarios. Face-to-face interviews are a common approach for such surveys and have been successfully used for inhabitants (Plieninger et al. 2013) and recreational visitors (Van Berkel and Verburg 2014; Zoderer at al. 2016). However, we doubted if it could be used for farmers, because they are notoriously busy. Therefore, we opted to survey the stakeholders' valuation of CESs by a website, which could be visited at any time that was most convenient for a respondent. The website was online for four weeks (October 5th till November 1st, 2015), during which we promoted it via newspapers, emails, the websites of the municipality and the water authority, and a stand at the annual "cow market" fair in Woerden, the main town near the research area. To encourage participation in the survey, the questionnaire was deliberately short, i.e., respondents could complete it within 10 minutes. We encouraged participation by offering respondents the chance to enter a prize draw in which the prize was a hot air balloon ride over the peatlands. 295 people responded to the survey. All stakeholder groups were well covered, with multiple respondents stating that they were both farmer and inhabitant, thus belonged to both stakeholder groups. The response covered approximately 5% of the adult population of the research area (sample frame 3,400), 58% of the farmers (sample frame 85), and 0.1% of the recreational visitors (sample frame 150,000). The coverage was slightly biased in terms of gender and age: 69% of the respondents were male, compared to 50% of the population of the research area. The average age of the respondents was 52 years, whereas the average age in the research area was 41 years.

The website contained a questionnaire to elicit (a) respondents' non-monetary valuation of the CES indicators, (b) their WTP for the scenarios, and (c) some of their general characteristics: their age, their gender, and the stakeholder group to which they belonged, i.e., farmers, inhabitants, or recreational visitors. The questionnaire was elucidated by several visualizations, because previous studies have shown that visualizations are a powerful tool to elicit stakeholders' preference for landscapes and their constituent elements (Dramstad et al. 2006; Van Berkel and Verburg 2014; Zoderer et al 2016). We followed up on these examples by using photographs of the research area to visualize the CES indicators and negative impacts (Fig. 4.4), and the current situation and the scenarios at three locations in the research area (Fig. 4.5). We used imaging software to manipulate photographs of the current situation in a way that reflected the developments envisioned in the scenarios.

Fig. 4.4. Visualizations of indicators and negative impacts for cultural ecosystem services.

In addition to the visualizations, all CESs were elucidated by short descriptions, e.g., for old farmsteads: "In the linear settlement zones and the villages of Kanis and Kamerik, several old farmsteads with characteristic farmyards are present, dating from the 19th century." We asked two questions about the CES indicators. First, we asked the respondents to give all individual CES indicators a value on a seven-point scale ranging from -3 (very negative) to +3 (very positive), in response to the question: "How do you value the following peatland features?". Second, we asked them to quantify the relative weight of the four CESs, by distributing 100 points among the four CESs, in response to the question: "How important are the following groups of peatland features for your overall valuation of the peatlands?".

Fig. 4.5. Visualizations of the scenarios on three locations in the research area (see Fig. 4.3).

The scenarios were elucidated by the topographic maps (Fig. 4.3) and the visualizations (Fig. 4.5). The maps were accompanied by some brief background information. The visualizations were accompanied by short descriptions, e.g., for location 3 in scenario 2: "The recreation area at "het Oortjespad" has expanded, adding several luxurious bungalows.". We asked one question about the scenarios: "What is the annual amount you are willing to pay to support the landscape, the cultural heritage, and the ecology in the current situation and the following scenarios?". Note that we used layperson's terms, because we assumed most respondents were not familiar with the ES framework and its terms.

<u>Assessing impacts</u>

In the third step of our research approach, we assessed the spatially explicit physical impacts of the current situation and the scenarios. We used a GIS-based integrated modeling framework for peatland management (van Hardeveld et al. 2017) to assess the spatially explicit impacts of the scenarios. As input for the initial situation of each scenario, the modelling framework required maps of the topography (Fig. 4.3), the terrain elevation, the soil properties, the surface water levels, and the groundwater tables. As output for the chosen timeframe, it produced maps of the terrain elevation, the soil subsidence, the surface water levels, the groundwater tables, the agricultural crop yields, and the ecological quality of the terrestrial vegetation. In addition, it assessed the required number of weirs and embankments, the required maintenance of roads, sewers, and utility cables, the damage to old farmsteads and other real estate due to changes in groundwater tables, and the CO_2 emissions due to subsidence of the peat soils. Subsequently, we used the topographic maps and the terrain elevation maps to map all lines of sight from the main road and the main hiking trail in the area. In addition, we used the topographic maps and the groundwater table maps as input for habitat models (van Dijk et al. 2014) with which we mapped the density of breeding meadow birds.

Analyzing costs and benefits, and mapping ecosystem services

The fourth step of our research approach consisted of a distributed CBA, and the construction of maps of the CES valuation by the stakeholder groups. We used the CBA approach of Van Hardeveld et al. (2018) to derive the costs and benefits of the scenarios for the stakeholder groups involved in the management of Dutch peatlands. This CBA approach reflects a multi-criteria discussion of costs and benefits from multiple evaluative endpoints, rendering it well suited for the participatory valuation process of our case study. As input, the CBA approach required the results from the integrated modeling framework, the assumed features of the scenarios (see Fig. 4.3 and appendix 4A), and empirical financial data. As output, it assessed (a) the management costs of the water system, distributed to the water authority, (b) the management costs of roads and sewers, distributed to the municipality, (c) the management costs of utility cables, and the benefits from recreational visits, distributed to the businesses, (d) the costs of real estate damage, distributed to the inhabitants, (e) the benefits of agricultural production, distributed to farmers, and (f) the non-financial benefits of reduced CO_2 emissions, distributed to society at large. In addition, we also assessed the non-financial bequest and existence values for farmers, inhabitants, and recreational visitors, by multiplying their population by the average WTP they stated in the survey.

We used the DialogueMaps tool to construct maps of the overall CES valuation. We constructed the maps separately for the current situation and the three scenarios, and considering the three stakeholder groups (farmers, inhabitants, and recreational visitors). The mapping approach resembled the construction of value maps by Janssen et al. (2005), and landscape preference maps by Van Berkel and Verburg (2014). First, using the results from the integrated modeling framework and the assumed features of the scenarios, we constructed maps of all individual CES indicators and negative impacts. Second, we averaged the values for the individual CES indicators and negative impacts which were stated by the survey respondents and assigned these averages spatially explicitly to the maps of CES indicator and negative impacts. Third, for all four CESs, we aggregated the results of step two spatially explicitly and divided the aggregated result at each location by the number of individual CES indicators or negative impacts per CES. Forth, for all four CESs, we averaged the relative weights which were stated by the survey respondents and multiplied these averages spatially explicitly by the resulting maps of step three. We derived the overall CES valuation maps by aggregating the four CES valuation maps. In addition, the overall CES valuation maps were spatially aggregated and adjusted for land uses that were not valued, i.e., built-up areas and cranberry fields. The results were transformed into an index score, with the average aggregated valuation of the current situation equal to 100.

Deliberating the results

In the fifth step of our approach, we discussed all results in two workshops (November 16th and November 23rd, 2015) with 16 stakeholders, representing the three stakeholder groups of the CES valuation. Nine stakeholders who also participated in the second workshop during the first step of our approach represented the recreational visitors. In addition, we invited seven stakeholders who participated in the survey to represent the farmers and the inhabitants.

The workshops started with an overview of the results of the CBA and the CES valuation. Next, we let them use the DialogueMaps tool to allow them to interactively examine all maps, and to ponder on and interpret spatial patterns, differences between scenarios and stakeholder perspectives. For each scenario and each stakeholder perspective, the DialogueMaps tool contained the overall CES valuation map, as well as five thematic maps of combinations of individual CES indicators: (1) land use, including the CES indicators "area of meadows", "area of cultivated fields", "area of meadow with grazing cows", "area of osier beds", "area with opportunities for overnight stays", and the negative impact "number of modern livestock barns", (2) lines of sight, i.e., the CES indicator "area visible from the main road and the main hiking trail", (3) cultural heritage, consisting of the CES indicators "number of ditches which are part of medieval ditch patterns", "area which likely contains archeological remains", and "number of old farmsteads", (4) infrastructure, including the CES indicators, "number of hiking trails" and "number of cycling tracks", and the negative impact "length of roads with much agricultural traffic", and (5) the CES indicator "area with high ecological quality", which reflects the quality of terrestrial vegetation as well as the density of breeding meadow birds. The thematic maps were incorporated along with several pop-up windows with photographs, visualizations, and brief background information we used in the survey.

The second part of the workshops consisted of a semi-structured discussion about (a) their interpretation of the CES valuation, (b) the feasibility of the scenarios and their constituent features, and (c) their advice for integrated management options in the research area. We concluded the workshops with a plenary moment for each participant to express their opinion about the research and the management of the research area. In addition, we asked all participants to answer three open questions in writing, i.e., what their perceived strong and weak points of our approach were, and what advice they could give us for further applications. Afterwards, we grouped their 124 expressed opinions and written answers into 13 feedback themes.

4.4 Results

Survey of stakeholders' valuation

The respondents tended to value the CES indicators positively (Table 4.2). The indicators that were valued most were "area of meadows", "area of meadows with grazing cows", "number of ditches which are part of medieval ditch patterns", and "area visible from the main road and the main hiking trail". Only the CES indicator "area which likely contains archeological remains" and the negative impact "length of roads with much agricultural traffic" tended to be valued negatively by some stakeholder groups. Remarkably, "number of modern livestock barns" was valued moderately positively, even though it negatively impacts the ensemble of cultural heritage values. The CES that contributed most to the overall valuation of the peatlands was "sense of place"; "recreation and ecotourism" contributed the least (Table 4.3).

Table 4.2. Average valuation of cultural ecosystem service indicators and negative impacts by stakeholder groups. Value scale ranges from -3 (very negative) to +3 (very positive).

Cultural ecosystem service	Indicators and negative impacts	Farmers N = 49	Inhabitants N = 177	Visitors N = 99	Total N = 295
Sense of place	Area of meadows	3.0	2.8	2.5	2.7
	Area of cultivated fields	1.3	0.8	0.3	0.6
	Area of meadows with grazing cows	2.8	2.6	2.0	2.4
Cultural heritage values	Number of ditches which are part of medieval ditch patterns	2.0	2.5	2.6	2.5
	Number of old farmsteads	1.2	2.0	2.3	2.1
	Area which likely contains archeological remains	-0.7	0.4	0.6	0.4
	Number of modern livestock barns	2.0	0.9	0.5	0.8
Aesthetic values	Area visible from the main road and the main hiking trail	2.3	2.4	2.5	2.4
	Area of osier beds	1.0	1.7	1.8	1.7
	Area with high ecological quality	1.7	2.1	2.2	2.1
Recreation and ecotourism	Length of hiking trails	1.0	1.5	1.6	1.5
	Length of cycling tracks	1.2	1.8	1.8	1.7
	Area with opportunities for recreational overnight stays	0.5	0.5	0.1	0.3
	Length of roads with much agricultural traffic	0.0	-0.4	-0.9	-0.6

Table 4.3. Relative contribution of the cultural ecosystem service to the stakeholders' overall valuation of the peatlands.

Cultural ecosystem service	Farmers N = 49	Inhabitants N = 177	Visitors N = 99	Total N = 295
Sense of place	59%	36%	28%	33%
Cultural heritage values	13%	22%	24%	23%
Aesthetic values	16%	26%	29%	27%
Recreation and ecotourism	12%	16%	19%	17%

The differences between stakeholder groups were most pronounced regarding the CES "cultural heritage values", with farmers valuing the individual CES indicators of this CES markedly lower, and the negative impact "number of modern livestock barns" markedly higher. In addition, farmers also valued the other CESs markedly differently than the other stakeholder groups, with lower scores for "aesthetic values" and higher scores for "sense of place". They all awarded the individual CES indicator "area of meadows" the maximum score.

The average annual WTP for the CESs in the current situation was €17.19 (Table 4.4). Remarkably, for each stakeholder group the average WTP is lower for all scenarios except for scenario 3 (wetter conditions), for which the recreational visitors are willing to pay more. Scenario 2 (scale enlargement of current land uses) is regarded most negatively, with the average WTP being 81% lower compared with the current situation. The differences between the scenarios can be attributed to differences in the amounts stated by the respondents who are willing to pay something, and inter-group differences in the number of respondents who are willing to pay something. For the current situation, 74% of all respondents are willing to pay €23.67 on average per annum, resulting in an average WTP of €18.53. For scenario 3 (wetter conditions) the average amount respondents are willing to pay is similar, i.e., €23.24 per annum, but because only 61% of the respondents are willing to pay something, the resulting average WTP is €13.91. For scenario 2 (scale enlargement of current land uses) the number of respondents willing to pay something is lower, i.e., 29%, and so is the average amount that they are willing to pay, i.e., €12.49, resulting in an average WTP of only €3.31 per annum.

Table 4.4. Stakeholder groups' average willingness to pay for the current situation and three scenarios [€ yr-1], and difference (in %) by comparison with willingness to pay for the current situation.

Scenario	Farmers N = 49		Inhabitants N = 177		Visitors N = 99		Total N = 295	
0. Current situation	€ 18.53		€ 19.38		€ 13.44		€ 17.19	
1. Business as usual	€ 17.68	-5%	€ 18.13	-6%	€ 13.08	-3%	€ 16.13	-6%
2. Scale enlargement of current land uses	€ 8.30	-55%	€ 2.93	-85%	€ 2.25	-83%	€ 3.31	-81%
3. Wetter conditions	€ 3.35	-85%	€ 10.86	-44%	€ 19.94	48%	€ 13.91	-19%

<u>Assessment of ecosystem services</u>

All the scenarios have markedly different impacts (see appendix 4A). For instance, scenario 2 (scale enlargement of current land uses) results in fewer grazing cows and fewer ditches. Because both these CES indicators are highly valued (Table 4.2), scenario 2 is valued less than the other scenarios. In addition, the valuation of the scenarios reflects the spatial patterns of the CES indicators. For instance, the landscape is very open in the west of the area but is much less open in the southeast, which is why the western part of the research area is valued more positively, and the southeastern part more negatively (Fig. 4.4).

Table 4.5. Index of the aggregated valuation of cultural ecosystem services by stakeholder groups for the current situation and three land use and water management scenarios (average valuation of the current situation = 100), and difference compared with the current situation [%].

Scenario	Farmers N = 49		Inhabitants N = 177		Visitors N = 99		Total N = 295	
0. Current situation	111		98		91		100	
1. Business as usual	94	-17%	86	-12%	85	-6%	89	-11%
2. Scale enlargement of current land uses	74	-37%	74	-24%	74	-17%	74	-26%
3. Wetter conditions	100	-10%	90	-8%	84	-8%	91	-9%

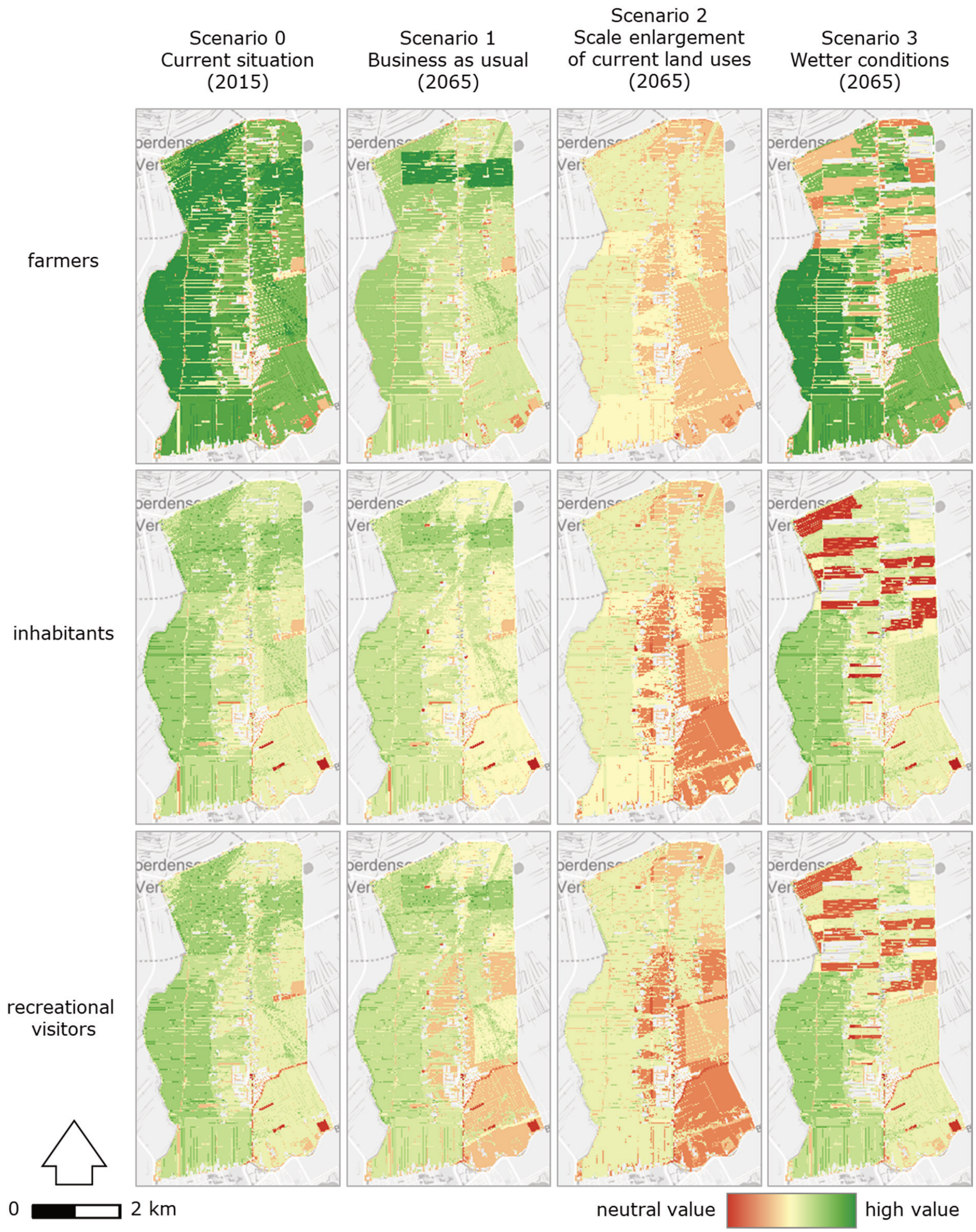

Fig. 4.4. Valuation of cultural ecosystem services by three stakeholder groups for the current situation and three scenarios. Note that the color range was chosen to accentuate differences, with red colors reflecting neutral values. Respondents were not asked to value built-up areas and cranberry fields.

The CES valuation maps (Fig. 4.4) present a different perspective on CES valuation than the WTP, with less pronounced differences between the scenarios and the stakeholders. The main reason is that the CES valuation is more comprehensive and spatially detailed than the single-value, non-spatial WTP amounts. For instance, the recreational visitors are willing to pay 48% more for scenario 3 (wetter conditions) than for the current situation

(Table 4.4), whereas their CES valuation index for scenario 3 is 8 percent lower (Table 4.5). The reason is they disfavor cultivated fields, which are abundant in scenario 3. Moreover, the cultivated fields negatively impact several of their favored CES indicators, because meadows with grazing cows and ecologically valuable meadow birds are displaced, and the openness is obstructed. Note that farmers have higher CES valuation indexes than the other stakeholders (Table 4.5). The main reason is the importance they attribute to the CES "sense of place", which has a major impact on all scenarios.

The scenarios not only impact the CES indicators but also the costs and benefits (Table 4.6). The main differences are caused by the WTP amounts, the water management costs, and the crop yields (see appendix 4A). Scenario 2 (scale enlargement of current land uses) has the most positive financial effects, but also the most negative non-financial effects. The reverse applies to scenario 3 (wetter conditions). The relative differences between the non-financial effects are much more pronounced than the differences in WTP (Table 4.4). Because the population of farmers is relatively small, their cumulative WTP is a modest sum for all scenarios, whereas the WTP of recreational visitors can be substantial, because their population is relatively large (see appendix 4A).

Table 4.6. Distributed monetary effects of three scenarios, i.e., the differences between the scenarios in 2065 and the current situation [€ ha-1 yr-1]. The financial and non-financial effects are aggregated separately.

Scenario	**1. Business as usual**		**2. Scale enlargement of current land uses**		**3. Wetter conditions**	
Effect	**Financial**	**Non-financial**	**Financial**	**Non-financial**	**Financial**	**Non-financial**
Water authority	€ -281		€ -29		€ -268	
Municipality	€ 42		€ 25		€ 98	
Businesses	€ 21		€ 64		€ 43	
Farmers	€ -1	€ 0	€ 250	€ -1	€ -273	€ -1
Inhabitants	€ 274	€ -2	€ 289	€ -29	€ 264	€ -15
Recreational visitors		€ 59		€ -657		€ 656
Society at large		€ 47		€ 156		€ 111
Total	€ 56	€ 104	€ 599	€ -531	€ -135	€ 751

<u>Feedback from stakeholders</u>

In the workshops we witnessed how stakeholders used both the CES valuation and the monetary assessments when deliberating the feasibility of the scenarios and their constituent features. The DialogueMaps tool revealed differences in CES valuation between stakeholder groups (Table 4.5), as well as the spatial consequences of stakeholder preferences that had so far been inexplicit (Fig. 4.4). This increased the awareness of CES values, as well as options for synergies and trade-offs. For example, if scale enlargement of current land uses (scenario 2) were to become necessary for financial reasons (Table 4.6), the stakeholders suggested that these developments be confined to the southeastern corner of the research area, where CES valuation is lowest (Fig. 4.4). In areas with higher CES valuations, they envisioned agricultural enterprises with mixed features of all scenarios.

Remarkably, the negative impacts “number of big modern livestock barns” and “length of roads with much agricultural traffic” were key features of the deliberations. Regarding the big modern livestock barns, inhabitants and recreational visitors stressed the negative impact on the CES “cultural heritage values”, and farmers claimed it positively contributed to the CES “sense of place” of the agricultural peatlands. This instigated a discussion of how differences in transcendental values shaped their valuation of the research area and how these differences should be reflected in the land use and water management in the research area. Eventually, they all agreed the livestock barns would be necessary if the scale of agricultural farms was enlarged (scenario 2), but several adjustments would be feasible, which could minimize their negative impact on the ensemble of cultural heritage values. A similar debate ensued regarding the roads with much agricultural traffic, with farmers claiming that traffic by the inhabitants of the villages also had a negative impact on the CES “recreation and ecotourism”. Eventually, they all agreed that new hiking trails (scenarios 3) were preferable to a new road to accommodate the agricultural traffic (scenario 2).

The approach’s strong point that stakeholders mentioned most frequently in their feedback was the suitability of the approach to support a participatory planning process (Table 4.7). Their acclaim especially focused on the ability of the approach to support a discussion between different stakeholder groups, and the awareness it raised for sustainable adaptations. The feedback on weak points mainly concerned disputable assumptions and shortcomings of DialogueMaps. It is noteworthy that the broad-brush character of the reconnoitering phase of our approach inevitably results in disputable assumptions, primarily because aspects of CESs are omitted or oversimplified. Both shortcomings can easily be remedied in subsequent phases that focus on secondary, more detailed and critical analyses and dialogues. For example, by including minority views of subgroups of stakeholders.

Table 4.7. Feedback from the 16 stakeholder participants in the workshops. The percentage of stakeholders that mentioned the feedback is given in parentheses.

Strong points of approach	• Well suited to support a participatory planning process (50%) • Enhances the apprehension of the social–ecological system (31%) • Quantitative non-monetary interpretation of values (31%) • User friendliness of DialogueMaps (31%) • Integrated assessment (13%)
Weak points of approach	• Several disputable assumptions (69%) • Several shortcomings of DialogueMaps (63%) • Difficult interpretation of the valuation maps (44%) • Arbitrary selection of stakeholder groups (38%)
Suggested advice	• Follow up process with more detailed assessments (94%) • Include additional assessments (63%) • Continue the quest for sustainable land use scenarios (56%) • Improve the technological performance of DialogueMaps (38%)

Almost all the stakeholders advised proceeding with more detailed assessments, believing this would further enrich the social learning process they had embarked on. Moreover, they suggested a further exploration of the underexposed features of the scenarios. Due to unfamiliarity with some CES indicators, some values appear to be not fully preformed, e.g., the value of archeological remains. The elicitation of these values may be enhanced by providing additional information prior to the valuation, e.g., an oral presentation by an expert on the subject.

4.5 Discussion

To further advance techniques for plural valuation of ESs, many authors have advocated the combined use of monetary and non-monetary assessments (Beria et al. 2012; Daniel et al. 2012; Guerry et al. 2015; van den Belt and Stevens 2016; Jacobs et al. 2016). Arguably, for a successful combination of assessments the mapping of CESs is key. Willemen et al. (2015) suggest best ES mapping practices should be robust, transparent, and stakeholder-relevant. However, most examples of CES mapping practices reflect research on a regional, national, or continental scale, frequently with a marked emphasis on recreation and ecotourism (Hernández-Morcillo et al. 2013).

Our case study reflects research on a local scale, with a focus on a broad range of CESs. We have demonstrated how in these settings, robust, transparent, and stakeholder-relevant CES valuation maps can be used to evaluate planning scenarios. In the workshops, we witnessed how stakeholders used the maps effectively in conjunction with monetary assessments and negotiated planning options across the preferences of multiple stakeholders and multiple evaluative endpoints. We therefore believe the added value of our case study is that it demonstrates how participatory CES valuation approaches (e.g., Dramstad et al. 2006; Van Berkel and Verburg 2014; Zoderer et al 2016) can be used to integrate CESs as an additional evaluative endpoint in multi-criteria approaches that combine monetary and monetary assessments (e.g., Newton et al. 2012; Bateman et al. 2013; Sijtsma et al. 2013). Moreover, it demonstrates how the CES valuation can engage the local stakeholders in a planning process, instigating a process of value deliberation and participatory design of management options.

Our approach can be of added value for other local planning processes too, provided that its strengths and weaknesses are duly considered. A main challenge of our approach is the difficulty of designing a set of management scenarios and CES indicators that can not only engage a broad population of stakeholders but can also yield credible valuation results. To achieve this, we solicited the help of local experts and governmental organizations. We suggest future research follows up this approach, exploring its strengths and weaknesses in detail. For example, the participatory selection of CES indicators will enable accurate targeting of the CESs most relevant for the key stakeholder groups. However, it also brings the challenge of integrating laypersons' perceptions of "their" social–ecological system with a scientific system of classifying ESs. In previous studies, this challenge was dealt with by incorporating additional indicators which did not reflect CESs (Saarikoski et al. 2018). In our case study, the participating stakeholders regarded "modern livestock barns" and "roads with much agricultural traffic" as key elements of a discussion on planning options. Yet these elements cannot be classified as CES indicators because they do not contribute positively to a CES. Neither are they ecosystem disservices because they lack an ecosystem origin (Shackleton et al. 2016). For these reasons, we opted to classify them as big negative impacts on CESs. Although this resulted in a survey that is easily interpretable from a layperson's perspective, methodologically it remains somewhat ambiguous. Further examples that address this issue are welcome.

Another challenge of our approach is the difficulty of designing a questionnaire which can objectively elicit the stakeholders' valuations. We used visualizations to illustrate our CES indicators (Fig. 4.3) and scenarios (Fig. 4.5). Although previous studies have shown that visualizations are a powerful tool to elicit stakeholders' preference for landscapes and their constituent elements (Dramstad et al. 2006; Van Berkel and Verburg 2014; Zoderer et al 2016), an evaluative bias can occur if respondents are unclear what exactly they are asked to evaluate (Scott and Canter, 1997). We tried to limit the bias by using non-manipulated photos of the research area to visualize the CES indicators, supplemented by short descriptions to elucidate what the photographs represented. We also used photographs of the research area and short descriptions to visualize the scenarios, but for obvious reasons, we needed to manipulate the photographs which represented the scenarios. To limit the bias that may have resulted from our manipulations, we chose to visualize multiple locations. Arguably, this may have been insufficient to exclude all bias in the stated WTP amounts. We are aware of the possible bias but deemed it acceptable for the broad-brush character of the reconnoitering phase of our case study, envisioning more comprehensive and accurate valuations would ensue in follow-up phases, including ample opportunities to deliberate to what extent shared values transcend the stated individual preferences.

Besides exploring how CES valuation can advance participatory resource management, our paper also aimed to point out avenues for effectively engaging stakeholders. In our case study, we achieved stakeholder involvement via a participatorily designed valuation strategy, a well-advertised internet survey, and deliberative workshops. The response of the three separate stakeholder groups in our survey was equal to or greater than the overall response in comparable CES valuation studies that employed more time-consuming face-to-face survey methods (Bryan et al. 2010; Plieninger et al. 2013; van Berkel and Verburg 2014). Moreover, our strategy resulted in a response of approximately 58% of the key stakeholder group in the research area, i.e. the farmers. We believe the high response rate reflects the accurate targeting of the CESs relevant for this stakeholder group. That farmers perceived the CESs as relevant was illustrated by the spontaneous urging of young farmers during our survey at the annual "cow market" fair to engage them in planning processes concerning "their" area. The commitment of farmers was reflected in the high values they attributed to the CES "sense of place" and its constituent indicators (Table 4.2). A contributing factor to the high response rate was the round-the-clock opportunity to respond to the survey: 60% of the farmers responded after working hours, in the evening or at night. We suggest both factors are duly considered when designing CES valuation approaches.

The main shortcoming of our case study was that for budgetary reasons the follow-up phase had to be postponed. We were therefore unable to further support and enrich the social learning process the stakeholders of our case study had embarked upon. In this regard, our case study shared the fate of many research-driven processes that were able to raise awareness of multiple ESs but were not used instrumentally to effectuate actual changes in resource management (Saarikoski et al. 2018). In our case study, the first loop of the social learning process stressed previous observations of how participatory mapping might elicit values not reflected in WTP amounts (Kenter 2016b). In our case study, the average WTP of the stakeholder groups suggested marked differences between the scenarios (Table 4.4). These results were largely caused by differences in the proportion of respondents unwilling to pay anything. Yet the CES valuations clearly showed that all respondents assigned some value to the CESs, resulting in less marked differences between the scenarios (Table 4.5). In the workshops we witnessed how this realization dawned on the participating stakeholders, instigating a process of value deliberation between stakeholder groups. Moreover, they discussed how differences in transcendental values shaped their valuation of the research area, e.g., regarding the reasons why farmers attributed higher values to the CES "sense of place" in comparison to other stakeholder groups.

It is plausible that a second loop of the social learning process that the stakeholders of our case study had embarked upon could have resulted in the formation of shared cultural values, in the sense that Irvine et al. (2016) conceive of such values as normative constructs reflecting communal values beyond the aggregated utilities of individuals. Such multi-loop social learning processes would require additional deliberative methods to elicit shared values. Kenter (2016a) presents a variety of approaches and methods for this task, ranging from analytical-deliberative—e.g., deliberative monetary valuation and participatory modeling—to interpretive-deliberative—e.g., storytelling and arts-led dialog. We believe our approach is well suited for integration with additional analytical-deliberative methods, provided that the interactive features of the DialogueMaps tool are further enhanced. For instance, the transparency of the valuation process may be enhanced by allowing participants to adjust their valuation scores (Table 4.2), which are currently regarded as fixed parameters. This option would also allow them to further deliberate their valuation, both individually and as a group. Furthermore, an interactive design of the scenarios would enhance the support to mediation and negotiation. This option would require the use of serious gaming techniques, transforming DialogueMaps into a tool for interactive simulation.

4.6 Conclusion

To advance our understanding of which valuation approaches are best suited to support decision-making in different contexts and at different scales, this paper has demonstrated how participatory CES valuation was used in the reconnoitering phase of a local planning process in a part of the Dutch peatlands, integrating monetary and social valuation techniques, underpinned by biophysical modeling. The added value of our approach is that it:

- successfully engages key stakeholder groups, by a participatorily designed valuation strategy, a well-advertised internet survey, and deliberative workshops;
- presents spatially explicit CES valuation maps from multiple stakeholder perspectives, revealing the spatial consequences of preferences that were not explicit beforehand;
- combines monetary assessments with non-monetary CES valuation, supported by the interactive mapping tool DialogueMaps, which can instigate a process of value deliberation between stakeholder groups and allows for negotiation across multiple evaluative endpoints.

Our approach can be of added value for other local planning processes too, especially in conjunction with additional deliberative methods. Ultimately, this may result in integrated valuation processes that enable multiple stakeholder perspectives and plural value dimensions to be integrated into policy and management decisions.

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Appendix 4A. Characteristics of the scenarios

In the final workshop of the first step of our research approach (see section 4.3), we used the features that were highlighted by the participants of a previous workshop to design coherent topographic maps of the current situation and three land use and water management scenarios with diverging impacts on the research area (Fig. 4.3). In addition, we made assumptions regarding the features of the scenarios (Table 4A1). In the third step of our research approach (see section 4.3), using the assumed characteristics and the maps, an integrated assessment was made of soil subsidence, and the resulting impacts (Table 4A2).

Table 4A1. Assumed features of the current situation and three scenarios.

Features	1. Business as usual	2. Scale enlargement of current land uses	3. Wetter conditions
Surface water levels	55 cm below ground in all agricultural areas, 35 cm below ground in villages	55 cm below ground in all agricultural areas, 35 cm below ground in villages	55 cm below ground in southern agricultural areas, 35 cm below ground elsewhere
Meadows	2,067 ha	2,075 ha	993 ha
Grazing cows	On all meadows	On meadows close to farmsteads	On all meadows
Cultivated fields	23 ha	6 ha	1,036 ha
Average farm size	65 ha	416 ha	51 ha
Farmers	66	21	83
Field drains	No	On all agricultural land	No
Medieval ditch patterns	Identical to the current situation	50% less ditches than in the current situation	Identical to the current situation
Modern livestock barns	5 large livestock barns	5 very large livestock barns	No large livestock barns
Osier beds	Identical to the current situation	Identical to the current situation	Several additional locations
Hiking trails	Identical to the current situation	Identical to the current situation	Additional hiking trail
Cycling tracks	Additional bicycle track	Identical to the current situation	Identical to the current situation
Opportunities for overnight stays	Recreational area expands by 2 ha	Recreational area expands by 5 ha	Recreational area expands by 2 ha
Traffic	Identical to the current situation	More spread out, due to an additional road	Identical to the current situation
Recreational visitors	165,000 yr^{-1} (+10%)	195,000 yr^{-1} (+30%)	180,000 yr^{-1} (+20%)
Villages	Identical to the current situation	Villages expand by 5 ha	Identical to the current situation

Table 4A2. Assessed features of the current situation and three scenarios.

Features	1. Business as usual	2. Scale enlargement of current land uses	3. Wetter conditions
Soil subsidence	8 mm yr^{-1}	6 mm yr^{-1}	7 mm yr^{-1}
CO_2 emission	45.5 10^6 kg yr^{-1}	32.5 10^6 kg yr^{-1}	37.9 10^6 kg yr^{-1}
Pumps	4	3	3
Weirs	51	47	52
Inlets	30	30	50
Embankments	33.8 km	14.4 km	32.5 km
Average height of embankments	99 cm	85 cm	91 cm
Houses with damaged foundations	214	214	212
Houses with damage due to high groundwater tables	270	186	485
Average crop yield on meadows	88%	94%	90%
Average crop yield on cultivated fields	99%	94%	99%

Chapter 5

How interactive simulations can improve the support of environmental management – Lessons from the Dutch peatlands

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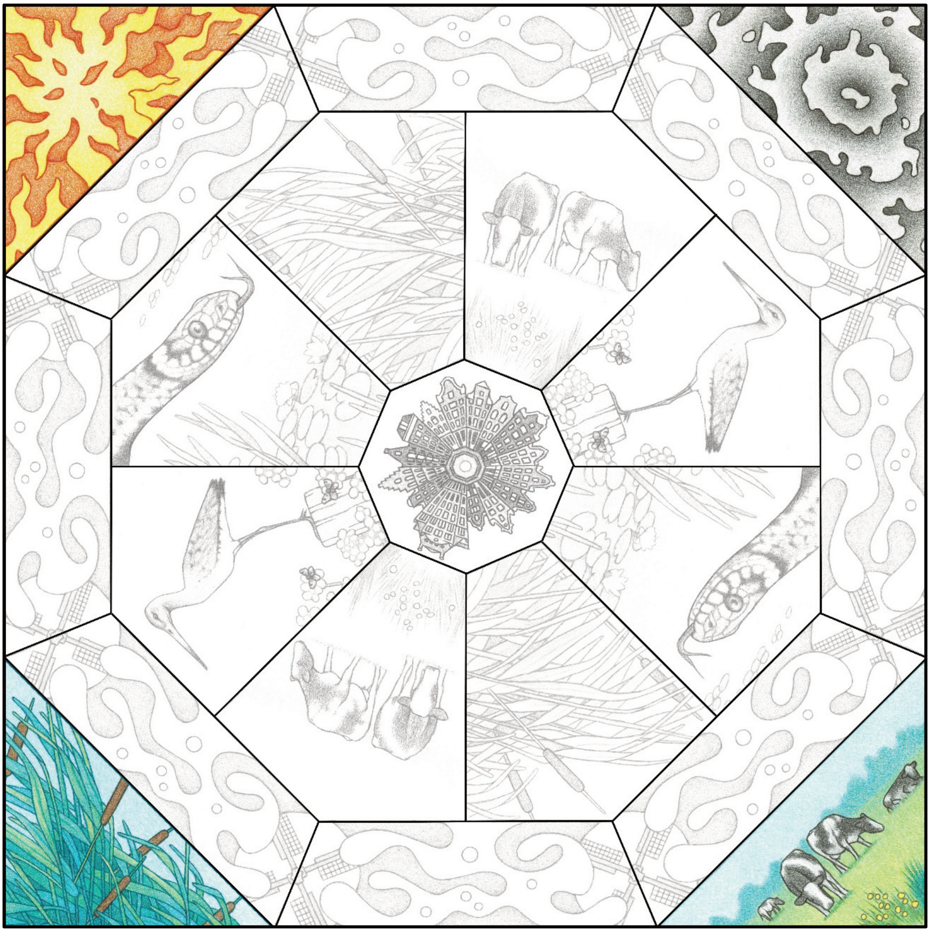

The long-term management options for peatlands are defined by a myriad of parameters. Four of them are depicted at the corners of the figure: the rate of climate change (top left corner), the regulations regarding greenhouse gas emissions (top right corner), the market price for paludiculture products (bottom left corner), and the market price for dairy products (bottom right corner).

Abstract

How interactive simulation systems can improve the support of environmental management is not fully understood. We therefore cross-analyzed questionnaires with logfiles and videos of workshops in which an interactive simulation system for peatland management was applied, to derive an in-depth perspective of its added values. The workshop participants explored the physical system dynamics, implementing measures, and the social system dynamics, brokering deals with other stakeholders. The system enabled capacity building at individual and group level, through iterative exploration of possible measures. As a result, cooperation among the stakeholders was enhanced and their understanding of problems and action perspectives regarding the peatlands was increased. Interventions that stimulated deliberation during the workshops were shown to prevent individualistic strategies, and instead fostered cooperative attitudes. The embeddedness in preceding Science–Policy Interfaces enhanced the credibility and legitimacy of the system, whereas salience was strengthened by abundant detailed information, realistic visual quality, and short calculation times.

5.1 Introduction

The sustainable management of social–ecological systems is notoriously complex because management strategies must address a set of interrelated environmental, political, and economic variables, with impacts across multiple spatial and temporal scales that are often nonlinear and highly uncertain (Walker et al., 2002; Ostrom, 2009). It is therefore widely acknowledged that management strategies must move beyond panaceas, instead adopting a perspective that embraces complexity (Folke, 2006; Ostrom, 2007). To effectively harness science for this challenge, interfaces are needed that promote communication and translation between experts and decision-makers and enable mediation to avoid tradeoffs between the salience, credibility, and legitimacy of the scientific information (Cash et al., 2003). These Science–Policy Interfaces are essentially social processes with the aim of enriching decision making (Van den Hove, 2007); they encompass a variety of typologies, such as individual mediators, processes of participatory knowledge development, and boundary organizations (Van Enst et al., 2014).

To bridge the gap between science and policy, many Science–Policy Interfaces use "boundary objects", i.e., collaborative outputs that ''are both adaptable to different viewpoints and robust enough to maintain identity across them'' (Star and Griesemer, 1989). Examples range from GIS technology (Harvey and Chrisman, 1998) and simulation models (White et al., 2010) to multifaceted concepts like "ecosystem services" (Abson et al., 2014). Cash et al. (2003) suggest that collaborative efforts to produce boundary objects are likely to result in credible, legitimate, and salient information.

However, Van Enst et al. (2014) point out that Science–Policy Interfaces may encounter several interaction problems that diminish their effectiveness. In their paper they illustrate how operational misfits between the demand and supply of knowledge will reduce the salience of information, and how strategic production and/or use of knowledge will also negatively impact the credibility and legitimacy of knowledge. The strategical interaction problems mainly occur when the knowledge is uncertain and/or consensus on norms and values is lacking. The operational misfits occur more often. For example, Uran and Janssen (2003) describe how many Decision Support Systems failed to provide salient information for their users and were therefore not used as effective boundary objects. Leskens et al. (2014a) describe how simulation models for flood disaster management encountered similar predicaments, mainly because the models needed experts to run them and could not keep pace with the speed of interactions in the decision-making processes.

In an extended literature review, Mayer (2009) describes how operational misfits and the accompanying critiques stimulated many developers of simulation models to create more transparent and interactive models that are more likely to become effective boundary objects. He argues that serious games can be regarded as the most promising exponent of this new generation of computer-mediated support systems, because they are able to integrate the technical–physical and the social–political complexities of policy problems. In addition, serious gaming is known to be an effective technique for learning and retention (Hofstede et al., 2010; Connolly et al., 2012; Wouters et al., 2013; Cheng et al., 2017), with proven abilities to engage stakeholders and allow them to experience the complexity of collaborative management tasks (Bekebrede, 2010; Vervoort et al., 2014). Not surprisingly, serious games are increasingly being used to support the management of social–ecological systems (e.g., Van der Wal et al., 2016; Voinov et al., 2016; Craven et al., 2017). For similar reasons, many contemporary Decision Support Systems allow for interactive simulations too; they include spatial decision support tools such as "Touch

Tables" (Arciniegas et al., 2013; Eijkelboom and Janssen, 2013; Pelzer et al., 2016), and flood simulation models (Leskens et al., 2014b). For terminological clarity, in this paper we will refer to all interactive computer-mediated support systems as "interactive simulation systems" (ISS).

Despite the potential benefits of ISS, it is unclear to what extent they effectively support management decisions, because much ISS research is hampered by one or more limitations. First, many ISS are tested with the help of students instead of real-world stakeholders (e.g., Hummel et al., 2011; Poplin, 2012; Arciniegas et al., 2013; Schulze et al., 2015). This raises the issue of external validity: it remains uncertain to what extent these settings reflect real-world practices. Moreover, the ISS test results of students have been shown to differ significantly from those of professional stakeholders (Bekebrede et al., 2015). Second, most studies consider only one or a limited number of workshops. Although this might provide valuable results, it remains uncertain to what extent the results can be generalized to other circumstances. Third, most studies focus on opinions voiced by the participants in workshops in which the ISS was tested, without considering logfiles and/or video recordings of the workshops. The disadvantage of stated opinions is that answers can be biased, e.g., by socially preferred answers. Logfiles and/or video recordings lack these possible biases, because they reveal not what people say but how they actually behave in the workshops in which the ISS is tested.

In this paper, we report research that aimed to overcome the research limitations mentioned above. We tested an ISS with real-world stakeholders in multiple workshops, using questionnaires, as well as logfiles and video recordings of the workshops. The guiding research question was: How can ISS improve the support of environmental management?

5.2 Method

<u>Outline research</u>

The case we used for our research was the collective management of Dutch peatlands. At the turn of the century, it was suggested to raise the surface water levels, which would decrease the soil subsidence rates. Although profitable dairy farming would no longer be possible and large-scale transitions from dairy farming to nature restoration would be necessary, this disadvantage would be outweighed by a decrease of management costs (Van Brouwers-Haven and Lokker, 2010). However, projects aimed at a top-down implementation of this strategy met with resistance from agricultural stakeholders. A lock-in situation developed, which raised awareness that more effective stakeholder collaboration was needed to produce legitimate results and develop viable management strategies.

To aid this resolve, various processes of participatory knowledge development have been instigated (Van Brouwershaven and Lokker, 2010), a boundary organization for innovative peatland management was created and alternatives to a top-down mode of environmental governance were explored (Den Uyl and Driessen, 2015). Nevertheless, at most locations, the soil subsidence rates have remained unsustainably high. Although it has been shown that innovative applications of field drains can reduce soil subsidence and improve the conditions for all stakeholders, this requires (a) a clear understanding of their site-specific impacts, and (b) consensus on a fair distribution of their costs among the stakeholders (Van Hardeveld et al., 2018). These implementation challenges are not easily overcome, because site-specific collaborative management strategies to reduce soil subsidence have not become commonplace.

For this context, we developed RE:PEAT, an ISS for the collaborative management of peatlands which accurately assessed the site-specific impacts of management strategies and supported negotiation processes on goals, means, and implementation pathways. Next, we applied RE:PEAT in ten workshops, in which the participants faced the assignment of improving the future conditions of a specific site in the Dutch peatlands. All participants could influence the simulation by stakeholder-specific actions and transactions with other stakeholders. We used post-workshop questionnaires to enquire about the workshop participants' perceptions of the added values of RE:PEAT to overcome the aforementioned implementation challenges for collaborative management strategies. To reveal how the participants used RE:PEAT, we also enquired about their attitude and their strategies, and recorded the workshop proceedings on logfiles and video. In addition, we experimented with different workshop settings and analyzed how these settings influenced the outcomes of the workshops. The combined results of our experiment were used to derive an in-depth perspective on how ISS can improve the support of environmental management.

Developing RE:PEAT

Aided by several key experts on the Dutch peatlands, we developed an ISS for peatland management. The core of the ISS consisted of a spatially and temporally explicit modeling framework that simulates the interrelated dynamics of surface water levels, phreatic groundwater tables, and soil subsidence, as well as the ensuing effects on embankments and hydraulic structures, real estate, CO_2 emissions, and crop yield (Van Hardeveld et al., 2017). Following the Cost–Benefit Analysis approach of Van Hardeveld et al. (2018), we combined the modeling framework with empirical economic data, to simulate the investment sums and maintenance costs required for the water system, field drainage, real estate, gardens, and roads and sewers, as well as the Net Value Added of the agricultural production and the agricultural supply chain.

We combined the expanded modeling framework with the Tygron Geodesign Platform, an interactive software platform for accurate 3D modeling of spatial development projects (Warmerdam et al., 2006; Bekebrede, 2015). The combination with the Tygron Geodesign Platform transformed most scenario settings of the extended modeling framework, e.g., the drainage strategy or the land use, into actions that allowed users to influence the simulation. In addition, the Tygron Geodesign Platform allowed for monetary transactions during the simulation, as well as the levying of taxes.

As we wanted the resulting ISS to reflect the entire range of land uses in Dutch peatlands (i.e., dairy farming and other forms of agriculture, villages, and nature reserves), we expanded the ISS with several additional effects that we deemed relevant for these land uses. We used empirical data from water authorities so as to include the water supply required by drainage strategies, the amounts of dredged material (the numerous ditches and waterways in the Dutch landscape must be dredged regularly), and the amount of nutrients that drain to the water system due to soil subsidence and farm management. The water quality was included by comparing the simulated nutrient loads with threshold values for nutrient loads above which ditches become choked with duckweed. The threshold values were obtained by 1,638 runs of the PCDitch model (Janse and Van Puijenbroek, 1998), allowing for variations in (a) soil properties, (b) water depth, determined by the surface water level, and (c) water discharge.

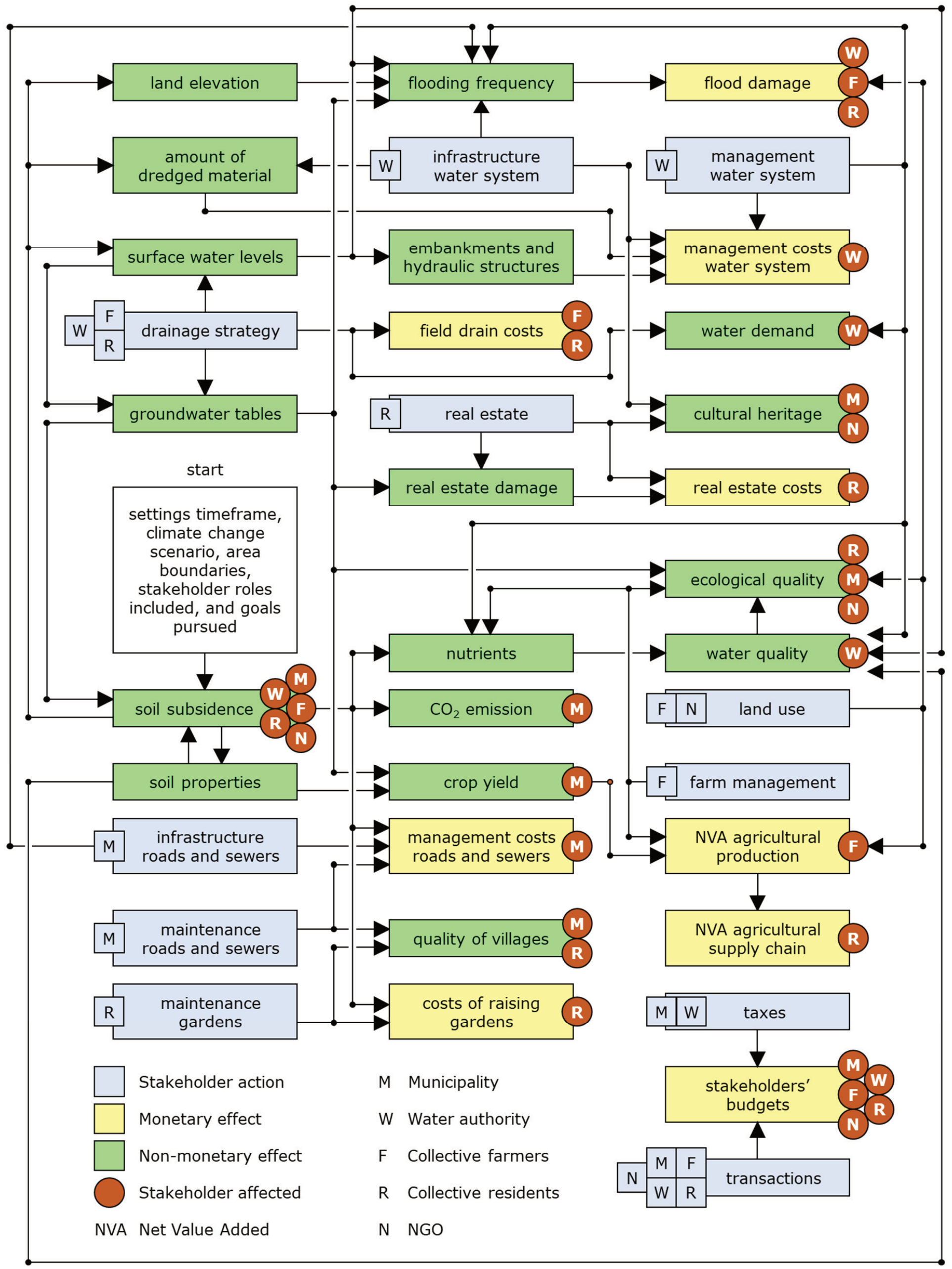

Fig. 5.1. Flowchart of the monetary and non-monetary effects simulated by RE:PEAT (adapted from Van Hardeveld et al., 2018). Arrows indicate the sequence of the simulations. RE:PEAT includes five stakeholder roles that can simultaneously implement actions that influence the simulation. The squares indicate which stakeholders can implement an action. The circles indicate which stakeholders are affected by an effect. For all these effects, goals are set for improving the current situation. Throughout the simulation, the extent to which the goals are achieved is monitored.

Drawing on the approach of Sijtsma et al. (2011), we included an ecological quality score derived from land use and groundwater tables, which we extended to reflect the effects of water quality and farm management too. We also included scores for urban quality and cultural heritage, which we derived from stakeholder actions. For example, demolishing old real estate diminished the cultural heritage score, and increasing the maintenance of gardens increased the village quality score. Flooding was included by using raster-based rainfall-runoff computations based on the diffusive wave approximation (Horritt and Bates, 2001), which adequately compared with the analytical solutions of overland flow presented by Di Giammarco et al. (1998), with Nash-Sutcliffe efficiencies exceeding 0.99.

The resulting ISS, called RE:PEAT (an acronym derived from Platform for Evaluating and Anticipating Trends in peatlands), can be used iteratively to explore the myriad of options in collaborative environmental management. It is shown in Fig. 5.1. We included five stakeholder roles in RE:PEAT: (1) the municipality, which manages the infrastructure of roads and sewers, (2) the water authority, which manages the water system, (3) the collective farmers, who own most of the rural area, (4) the collective residents, who own most of the real estate in the villages, and (5) an NGO which manages the nature reserves. Note that in reality, farmers and residents predominantly operate individually. However, because their individual stakes were similar, for the sake of clarity and effectiveness, they were included collectively.

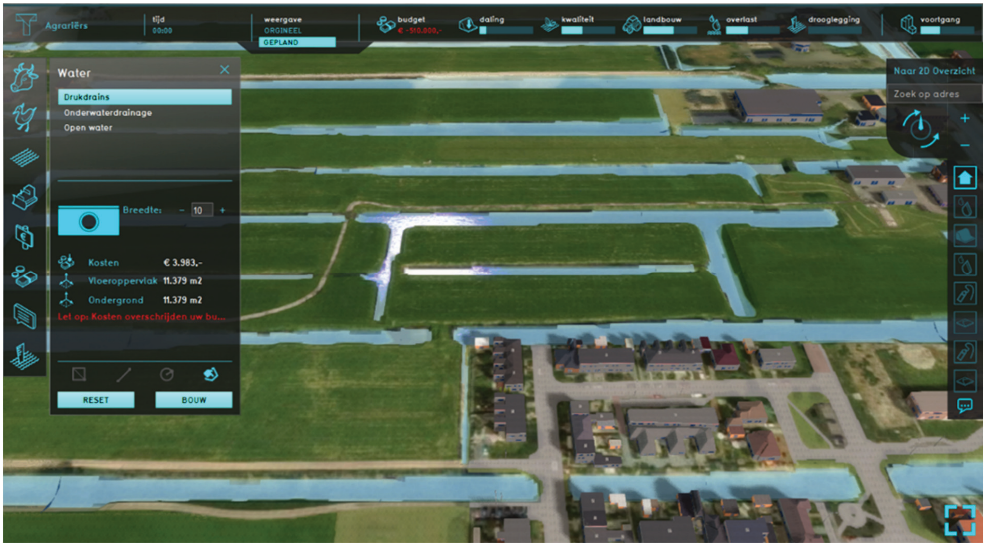

Fig. 5.2. Impression of the user interface of RE:PEAT. All users see the 3D simulation from their own perspective. The left bar contains possible actions, the right bar contains thematic maps, the top bar contains drop-down panels with information on goals, budgets, and a comparison of results with and without actions.

All stakeholder roles had a main individual goal and several accompanying goals: e.g., the main goal for the water authority was the reduction of management costs, with as accompanying goals the improvement of water quality and the reduction of water demand and flood damage. In addition, all stakeholder roles shared the common goal of reducing soil subsidence. Several stakeholder roles shared accompanying goals as well: e.g., improving the quality of the villages was important for both the municipality and the collective residents. All stakeholder roles had a personalized graphical user interface available, which contained action menus, thematic maps, information panels, and a view on the 3D simulation from their own perspective (Fig. 5.2). The information panels showed their budgets and the extent to which their goals were reached. For all goals, the panels compared the results with and without actions. In addition, an overall information panel showed a graph of their progress score throughout the simulation. The progress score was derived from the main individual goal (40%), the accompanying individual goals (30%), and the common goal of reducing soil subsidence (30%).

Comparing workshops

We organized ten workshops in which we applied RE:PEAT (from March 10th till November 29th, 2016). The workshops were attended by a total of 89 participants, who were professionally involved in the management of Dutch peatlands. In addition, each workshop was attended by a facilitator, who oversaw the overall workshop process, and by 2–4 assistants, who could provide technical support if needed. All workshops alternated rounds of interactive simulation with plenary moments of instruction and reflection. The time spent on these activities varied (Table 5.1). In general, the workshops started with 30–90 minutes of plenary instruction, followed by two rounds of interactive simulation which both lasted 30–45 minutes. The rounds of interactive simulation were followed by plenary debriefings, which lasted 5–10 minutes after round one and 15–25 minutes after round two. Drawing from the guidelines for debriefing of Peters and Vissers (2004) and Kriz (2010), the debriefing in-between rounds focused on the perceptions of the participants, a joint reconstruction of what happened, and a discussion of further options for actions. The final debriefing addressed the connection between the simulation and reality, including speculation about hypothetical scenarios and exploration of pathways to put into practice the lessons that were learned. On several occasions we deviated from the general approach. For example, the participants in workshops 3 and 4 opted to spend more time on plenary instruction, at the expense of interactive simulation time. In workshop 10, the available time was relatively limited, so we economized on the time allocated to instruction by assigning a technical assistant to each stakeholder role, to help the participants operate RE:PEAT.

Table 5.1. Settings of the workshops.

Workshop no.	**Time (hours:minutes)**			**No. of roles included**	**Style of the governmental roles**	**Participant involvement**
	Interactive simulation	**Plenary**	**Total**			
1	1:03	1:43	2:47	5	Top-down	Hands-on
2	1:06	1:50	2:56	5	Top-down	Hands-on
3	0:33	2:22	2:55	3	Top-down	Hands-on
4	0:33	2:22	2:55	3	Top-down	Hands-on
5	1:23	0:57	2:20	4	Deliberative	Hands-on
6	1:23	0:57	2:20	4	Deliberative	Hands-on
7	1:18	0:47	2:05	4	Deliberative	Hands-on
8	1:18	0:47	2:05	4	Deliberative	Hands-on
9	0:48	1:56	2:44	4	Deliberative	Hands-on
10	1:03	0:20	1:23	4	Deliberative	Guided

The settings of RE:PEAT reflected peatland areas of 9 km² and timeframes of 30–100 years, in which the gradual impacts of soil subsidence become apparent. For example, due to differences in soil subsidence rates, the differences in water levels between adjacent watercourses may increase. At some moment in time, this will require additional embankments to prevent the watercourses with higher water levels from slumping (Van Hardeveld et al., 2017). The exact moment in time that the embankments are needed depends on the characteristics of the peatland area. The chosen timeframes were always sufficiently long to include the moment at which such impacts were manifested in the peatland areas that were considered. To accurately assess the soil subsidence rates throughout the considered timeframes, we took into account that the microbes that oxidize peat become more active when the temperature rises (Tate, 1987). Therefore, we gradually adjusted the soil subsidence assessment, to reflect a regional projection of 2 °C global temperature rise (van den Hurk et al., 2006).

We included 3–5 stakeholder roles. In workshop 1–9, each stakeholder role was allocated to pairs of workshop participants, who shared a laptop computer. Workshop 10 was an exception, with 5–6 participants per stakeholder role. In this workshop, the laptops were connected to large projection screens, to assure that all participants could see the user interface (Fig. 5.2) during the entire workshop. Due to the limited availability of hardware, the NGO was only included in workshops 1 and 2. In workshops 3 and 4, the participants requested omitting the collective residents, so as to focus more on the remaining three stakeholder roles.

To examine how RE:PEAT can improve the support of environmental management, we experimented with the settings regarding the style of the governmental roles and the involvement of the workshop participants (Table 5.1). We used two styles of the governmental roles to examine the effect of interventions that stimulate deliberation. In workshops 1–4, we allowed the municipality and the water authority to make top-down decisions, i.e., they did not require other stakeholder roles to consent to changing taxes and drainage strategies. These workshops allowed for a top-down implementation of drainage strategies, similar to what was considered at the turn of the century in the Dutch peatlands. For workshops 5–10 we changed this set-up, forcing the governmental stakeholders to deliberate their decisions, i.e., they could only implement taxes and drainage strategies after obtaining the consent of the affected stakeholder roles. These workshops reflected the current ideas on peatland management, which acknowledge that cooperation between stakeholders is needed to produce viable management strategies.

We also experimented with the involvement of the workshop participants, to examine the effect of various application styles. Workshops 1–9 had a hands-on approach, with the participants operating RE:PEAT themselves. These workshops reflected a common setting of multi-player serious game sessions, which had not been used before to support the management of the Dutch peatlands. Workshop 10 had a guided approach, with the technical assistants operating RE:PEAT on behalf of the participants. These workshops reflected a common setting of touch table sessions, which had been used on several occasions to support the management of the Dutch peatlands before our experiment (Arciniegas et al., 2013; Brouns et al., 2015). Overall, this resulted in three groups of workshops with different settings: (1) workshops 1–4 had a top-down government style and hands-on workshop participants, (2) workshops 5–9 had a deliberative governmental style and hands-on workshop participants, and (3) workshop 10 had a deliberative governmental style and guided workshop participants.

Because the workshops varied regarding the number of stakeholder roles and the duration of the interactive simulation (Table 5.1), we performed a sensitivity analysis. First, we analyzed the sensitivity of the results to excluding stakeholder roles that were not included in all workshops, i.e., the "collective residents" not included in workshops 3 and 4, and the

"NGO" not included in workshops 3–10. Second, we analyzed the results' sensitivity to excluding workshops with less than one hour allocated to interactive simulation, i.e., workshops 3, 4, and 9.

Perceiving added values

We used post-workshop questionnaires to enquire about the workshop participants' perception of the added value of RE:PEAT to overcome implementation challenges of site-specific collaborative management strategies to reduce soil subsidence. In particular, we enquired about (a) enhancing cooperation among them, and (b) increasing their understanding of problems and action perspectives regarding the peatlands. We used five-point Likert scales to measure their perceptions, ranging from -2 (very negative) to 2 (very positive). We used pairwise Mann–Whitney tests to assess statistical differences between the three groups of workshops.

In addition, we included an open question in the post-workshop questionnaires, enquiring about arguments to elucidate the perceptions of added values. Afterwards, we classified the 166 responses about the perceived added values into six categories. We derived categories 1–4 from Pelzer et al. (2014), who distinguished between added values of ISS on (1) the individual level, regarding learning about the nature of the planning object, (2) the individual level, regarding learning about the perspective of other stakeholders, (3) the group level, i.e., the improvement of collaboration, communication, consensus, and efficiency, and (4) the outcome level, i.e., better-informed decisions. In a follow-up study, Pelzer et al. (2016) found that participants in ISS workshops perceived the added values at individual level to be key. We therefore selected both individual values as separate categories. In addition, we included categories for (5) the context of the application, e.g., the characteristics of the participants, the policy process, and the political context (Geertman, 2006), and (6) the usability of RE:PEAT, e.g., transparency, user friendliness, calculation time, and integrality (Pelzer et al., 2016). We used Fisher's exact tests to assess statistical differences between the three groups of workshops.

Exploring different uses

To reveal how the workshop participants used RE:PEAT, we used logfiles that recorded all the actions of the stakeholder roles during the simulation. In addition, we used multiple video cameras to capture the activities of the actual workshop participants too. Afterwards, we synchronized the videos and time-coded the activities of each workshop participant. Drawing on the system for coding group working relations developed by Nyerges et al. (2006), we used four codes to annotate the activities of the workshop participants: (1) inactive, e.g., checking a cell phone or pouring a glass of water, (2) reflective, i.e., (a) getting support from technical assistants, and (b) observing the interaction between other stakeholder roles, (3) interactive, i.e., discussion with other stakeholder roles, and (4) explorative, i.e., (a) focusing on the computer screen, and (b) discussion with participants within the same stakeholder role. For each participant, we logged the cumulative number of actions and interactions hour^{-1}, and the cumulative time spent on all coded activities. We also used the logfiles to examine to what extent the workshop participants reached their goals, and to what extent their own actions and the actions of other participants contributed to their overall progress score. We used pairwise Mann-Whitney tests to assess statistical differences between the groups of workshops. Regarding the time codes, we excluded the inactive episodes (time code 1), which on average accounted for 3.8% of the time.

We used post-workshop questionnaires to enquire about the participants' perception of their attitude. We used seven-point Likert scales to measure these perceptions, ranging from -3 (very uncooperative) to 3 (very cooperative). In addition, we included an open question, enquiring about the strategies they employed during the workshop. Afterwards, the 210 responses to the open question were grouped into five categories: (1) influencing the social system, e.g., brokering deals with other stakeholders, (2) influencing the physical system, e.g., implementing measures, (3) improving personal welfare, either by maximizing profits or by minimizing costs, (4) improving the peatlands, e.g., by minimizing the soil subsidence, and (5) no clear strategy. We used pairwise Mann–Whitney tests to assess statistical differences between the three groups of workshops regarding the attitudes of the workshop participants. To assess differences in their strategies, we used Fisher's exact tests.

5.3 Results

Differences in the perceived added values

In post-workshop questionnaires, the workshop participants clearly stated they perceived RE:PEAT to be of high added value for enhancing cooperation among them and increasing their understanding of the social–ecological system (Table 5.2). The perceptions of the groups were consistent, with only small differences between them. The proportion of workshop participants who elucidated their perceived added values with arguments regarding the outcome level was low; differences between the groups were not significant. Arguments regarding the added value at group level and individual level were more common. The argument "more awareness of other perspectives" was used by the largest proportion of workshop participants to explain their perception of the added values. The proportion using this argument differed significantly between the groups in workshops 1–4 and workshop 10 ($p=0.037$). The group of workshop 10 also stood out regarding the proportion that used the argument "improved understanding of the peatlands": it was significantly lower than the proportions for the groups of workshops 1–4 ($p=0.017$) and workshops 5–9 ($p=0.045$).

Table 5.2. The average added values perceived by groups of workshop participants for enhancing cooperation and increasing understanding, and the proportions of the groups that used an argument to explain their perceptions. The scale of added values ranges from -2 (very negative) to 2 (very positive). Significantly different results ($p<.05$) between the workshops 1–4, 5–9, and 10 are denoted by the letters a, b, and c.

Workshop group	1–4	5–9	10	All
No. of participants	32	35	22	89
Added value for enhancing cooperation	1.3	1.3	1.5	1.3
Added value for increasing understanding	1.2	1.3	1.0	1.2
More awareness of other perspectives	38% [c]	60%	77% [a]	55%
Improved understanding of the peatlands	58% [c]	50% [c]	15% [a,b]	46%
Support to the group process	42%	33%	15%	33%
Better-informed decisions	8%	3%	0%	4%
Context of the application	65% [c]	57% [c]	8% [a,b]	51%
Usability of RE:PEAT	50%	37%	46%	43%

Approximately half of the participants remarked that the added values strongly depend on the context of the application, such as the workshop setting and the characteristics of the participants. Some of them elucidated their remark by suggesting that the absence of conflicts was an important precondition for the added values. Their general perception was that although conflicts have not disappeared, there is a trend toward consensus and cooperation among the stakeholders in Dutch peatlands. Only for such contexts did they perceive high added values. Interestingly, the participants in workshop 10 seemed less troubled by such considerations: the proportion making such remarks was significantly smaller than in workshops 1–4 ($p=0.006$) and workshops 5–9 ($p=0.001$).

Almost half of the workshop participants mentioned that the usability of RE:PEAT contributed to their perception of the added values. Specifically, they mentioned the credible results, the abundance of detailed information, and the realistic visual quality of the user interface. For example, the impact of site-specific soil subsidence rates on the length of watercourses that required embankments to prevent them from slumping, or the impact of site-specific groundwater tables on the Net Value Added of dairy farms. Some of them acknowledged that in general they struggled to comprehend the full complexity of peatland management. They found RE:PEAT very useful because it presented a clear overview of all the aspects.

Behavioral differences during workshops

Logfiles and video recordings of the workshops revealed that on average, the workshop participants simulated 12.4 actions hour^{-1}, interacted with other participants 30.4 times hour^{-1}, and spent most of their time on exploration (Table 5.3). Per individual, the number of actions and interactions hour^{-1} differed markedly, with ranges of 2–38 actions hour^{-1} and 6–74 interactions with other participants hour^{-1} (Fig. 5.3). The average ratio of actions hour^{-1} to interactions hour^{-1} was 0.6, with only 16% of the participants exhibiting ratios greater than 1.0, i.e., engaging in more actions hour^{-1}. How the individual participants spent their time in the workshop also differed markedly, with ranges of 4–65% for time spent on reflection, 16–79% for time spent on exploration, and 5–52% on time spent interacting with other participants (Fig. 5.4).

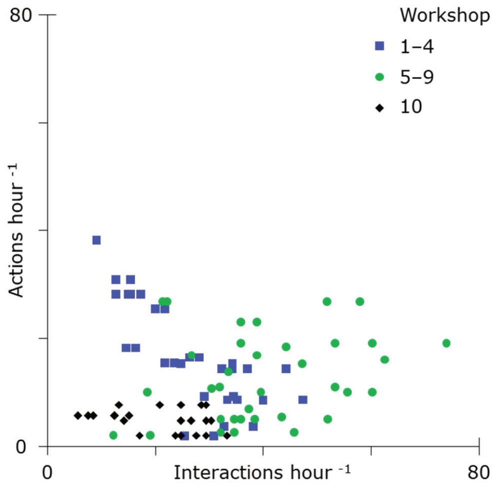

Fig. 5.3. The number of actions and interactions hour-1 of the workshop participants.

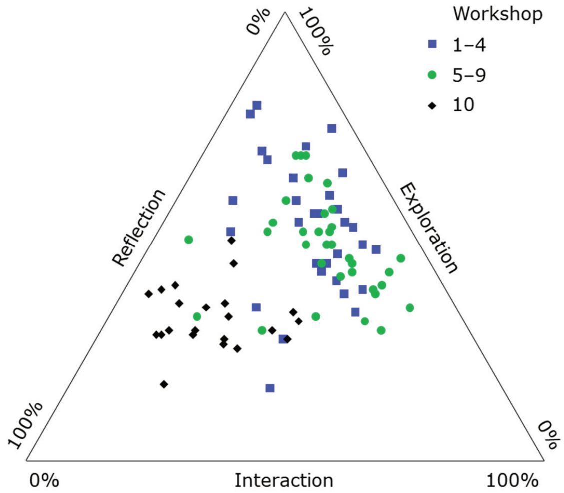

Fig. 5.4. The proportion of time spent by the workshop participants on reflection (left axis), exploration (right axis), and interaction (bottom axis).

Table 5.3. The average number of actions and interactions hour-1 by groups of workshop participants, and their average proportion of active time spent on exploration, on reflection, and on interaction. Significantly different results (p<.05) between workshops 1–4, 5–9, and 10 are denoted by the letters a, b, and c.

Workshop group	**1–4**	**5–9**	**10**	**All**
No. of participants	32	35	22	89
Actions hour^{-1}	17.3 ^{b,c}	12.8 ^{a,c}	4.9 ^{a,b}	12.4
Interactions hour^{-1}	25.9 ^{b}	40.4 ^{a,c}	20.9 ^{b}	30.4
Time spent on reflection	19% ^{c}	19% ^{c}	48% ^{a,b}	26%
Time spent on exploration	52% ^{c}	47% ^{c}	32% ^{a,b}	45%
Time spent on interaction	29% ^{c}	34% ^{c}	20% ^{a,b}	29%

To some extent, the variety in the behavior of the workshop participants related to the workshop settings. The participants in workshop 10 spent much time on dialog within their group with the technical assistants assigned to their stakeholder role. Therefore, they embarked on relatively few actions and interactions hour^{-1}. Consequently, their results differed statistically from both other workshop groups in terms of their average proportion of time spent on reflection (workshops 1–4 group: U=17.0, p=0.000; workshops 5–9 group: U=24.0, p=0.000), their average proportion of time spent on exploration (workshops 1–4 group: U=75.0, p=0.000; workshops 5–9 group: U=87.0, p=0.000), their average proportion of time spent on interaction (workshops 1–4 group: U=192.0, p=0.005; workshops 5–9 group: U=144.0, p=0.000), and their average number of actions hour^{-1} (workshops 1–4 group: U=84.0, p=0.003; workshops 5–9 group: U=31.0, p=0.000).

Regarding the number of interactions hour^{-1}, workshops 5–9 were statistically different (workshops 1–4 group: U=233.0, p=0.000; workshop 10 group: U=80.0, p=0.000). The governmental decisions in workshops 1–4 did not require consent from other stakeholder roles. Consequently, compared with the participants in workshops 5–9, the participants in workshops 1–4 had fewer interactions hour^{-1} and more actions hour^{-1} (U=258.0, p=0.030).

Although the governmental decisions in workshop 10 also required consent from other stakeholder roles, this did not result in markedly more interactions hour^{-1} than in workshops 1–4, because the participants in workshop 10 spent much time on discussions among the participants who shared their stakeholder role.

On average, the overall progress score during the workshops was 22% (Table 5.4). This may seem rather modest, but it must be noted that due to opposite effects of actions, high scores were very difficult to realize. For example, a raise in surface water levels would decrease the soil subsidence rate, which would increase the overall progress score. However, the frequency of flooding would increase as well, which would lower the overall progress score. Due to such opposite effects, only two pairs of workshop participants achieved an overall progress score of more than 50%. Remarkably, in workshops 5–10 the average progress was mainly caused by the actions of other participants, whereas in workshops 1–4 the progress was caused by the actions of the participants themselves. In this group of workshops, the progress due to the actions of other participants was significantly lower than in workshops 5–9 (U=83.0, p=0.014). The difference coincides with the significantly lower number of interactions hour^{-1} (Table 5.3)

Table 5.4. The average extent to which groups of workshop participants reached their goals, their average progress score, and the average extent to which their progress was caused by their own actions or the actions of other participants. Significantly different results (p<.05) between workshops 1–4, 5–9, and 10 are denoted by the letters a, b, and c.

Workshop group	1–4	5–9	10	All
No. of participants	32	35	22	89
Common goal: less soil subsidence	30%	46%	33%	38%
Individual goals	13%	17%	20%	16%
Overall progress score	18%	25%	26%	22%
Progress due to own actions	10%	4%	9%	7%
Progress due to actions of others	8% [b]	21% [a]	16%	15%

Post-workshop questionnaires revealed that the participants used four types of strategies (Table 5.5), on average to almost the same extent. Significant differences were only found regarding the proportion of strategies aiming to influence the physical system: this was lower in workshop 10 than in workshops 1–4 (p=0.008). The difference coincides with the significantly lower number of actions hour^{-1} (Table 5.3) and the significantly lower proportion of participants using "improved understanding of the peatlands" as an argument to explain their perception of added values (Table 5.2). It seems that the participants in workshop 10 focused more on the social system dynamics than on the physical system dynamics. Consequently, they might not have increased their understanding of the social–ecological system as comprehensively as the participants in the other workshops. However, their appreciation of the added value of RE:PEAT did not reflect this shortcoming, either because the effect was limited, or because they were unaware of it.

On comparing participants' attitude during the workshops, we found that for workshops 1–4, in which governmental stakeholders were able to enforce top-down decisions, scores were significantly lower than in workshops 5–9 (U=268.0, p=0.016) and workshop 10 (U=109.5, p=0.012), in which governmental stakeholders needed to deliberate their decisions with the other stakeholders. Note that on average, the lower scores did not reflect uncooperative impressions but impressions that were neutral to slightly cooperative. Uncooperative attitudes were only expressed by the participants in workshops 2–4. In these workshops, many participants behaved individualistically and exhibited ratios of actions hour^{-1} to interactions hour^{-1} of up to 4.2, i.e., strongly preferring individual actions over interaction and deliberated coordinated actions. Because in workshops 5–10 deliberation was mandatory for the governmental stakeholder roles, cooperative attitudes prevailed, with only one of the 67 participants expressing a slightly uncooperative attitude. It is noteworthy that participants in workshop 1 spontaneously engaged in several coordinated actions that required much deliberation, resulting in cooperative attitudes similar to workshops 5–10 (an average attitude score of 1.5).

Table 5.5. The proportion of groups of workshop participants that employed a strategy, and their average attitude. The scale of attitude ranges from -3 (very uncooperative) to 3 (very cooperative). Significantly different results (p<.05) between workshops 1–4, 5–9, and 10 are denoted by the letters a, b, and c.

Workshop group	1–4	5–9	10	All
No. of participants	32	35	22	89
Influence the social system	52%	42%	41%	45%
Influence the physical system	76% [c]	39%	12% [a]	45%
Improve the personal welfare	32%	55%	35%	43%
Improve the peatlands	44%	33%	41%	39%
No strategy	4%	6%	12%	7%
Attitude	0.4 [b,c]	1.3 [a]	1.5 [a]	1.0

Sensitivities analyzed

The sensitivity analysis (see Appendix 5A) revealed only minor changes in the results, due to the exclusion of (a) the stakeholder roles that were not included in all workshops, i.e., "collective residents" and "NGO", and (b) the workshops with less than one hour allocated to interactive simulations, i.e., workshops 3, 4, and 9. Any changes in statistically significant differences between groups of workshops primarily reflected the smaller group sizes resulting from the exclusions. We therefore deem the results not biased by variations in the workshop settings.

5.4 Discussion

System design

The system design of an ISS determines to what extent it can be used as a boundary object for the management of social–ecological systems. The general design principles are that the ISS should always promote communication and translation between experts, and promote mediation to avoid tradeoffs among the salience, credibility, and legitimacy of the scientific information (Cash et al., 2003). In our project, we aimed to secure these functions at the developmental stage of RE:PEAT by recruiting several key Dutch peatland experts to translate existing scientific knowledge into content that was salient from a stakeholder perspective, understandable by non-scientific participants, yet scientifically credible. Much of the knowledge incorporated in RE:PEAT resulted from preceding Science–Policy Interfaces, such as processes of participatory knowledge development (Van Brouwershaven and Lokker, 2010) and a boundary organization for innovative peatland management. We believe the embeddedness of RE:PEAT in preceding Science–Policy Interfaces was an important condition for credibility and legitimacy. Arguably, without this embeddedness, the incorporated knowledge would have been more uncertain and disputable, which would have diminished the effectiveness of the ISS.

The workshop participants mentioned that the good usability of RE:PEAT also resulted from the abundance of detailed information and the realistic visual quality of the user interface. Arguably, these features enhanced the salience of the information for them. Throughout the simulation, they were continuously presented with sufficient information to make decisions, in an easily understandable format. The extent to which these ISS features can be enhanced is related to the resulting calculation times. An important condition for ISS is its ability to keep pace with stakeholder interactions during actual decision-making processes (Eijkelboom and Janssen, 2013; Leskens et al., 2014a). In our case, we were able to enhance the information load and the visual quality of RE:PEAT quite extensively, because the graphics processing unit was an integral part of its computing system. Therefore, the maximum calculation times were limited to a few seconds per action.

The ISS features that enhance its usability are strongly related to the main added values that were perceived by the workshop participants. Like Pelzer et al. (2016), we found that ISS workshop participants perceived the added values at individual and group level to be key, and the added value of a better-informed outcome to be less important. In the workshops, we witnessed how the participants collaboratively designed adaptive drainage strategies that could slow down soil subsidence. Although the generic effects of these adaptive drainage strategies had been known for approximately a decade (Querner et al., 2012), they had never been implemented on a large scale. Because RE:PEAT translated the generic scientific knowledge into site-specific effects from multiple stakeholder perspectives, it created an operational fit between knowledge supply and demand. In addition, because the RE:PEAT supported informed negotiations, it also raised awareness of mutually beneficial strategies. It is noteworthy that later, several workshop participants initiated a collaborative process to implement the adaptive drainage strategy designed during the workshops in the same peatland area they explored with RE:PEAT. Moreover, they intend to continue using RE:PEAT as a boundary object, to support collaborative management decisions in the years to come.

We believe the results show that the general purpose of ISS should reflect capacity building at the individual and group level, striving for a site-specific awareness of the effects of measures and strengthening the resolve of the stakeholders to collectively implement these measures. ISS applications should not primarily focus on better-informed outcomes. Although credible results are obviously important, the iterative and interactive exploration of the myriad of management options should be the key consideration in support systems for collaborative environmental management.

<u>Workshop setting</u>

Our experiment with different workshop settings revealed that all workshop participants explored the physical system dynamics, implementing measures, and the social system dynamics, brokering deals with other stakeholders. However, the participants in the workshops with a top-down government style implemented markedly more actions hour^{-1} than the participants in the other workshops. In addition, their strategies were primarily aimed at influencing the physical system, and their attitude during the workshops was significantly less cooperative than that of the participants in the other workshops. Consequently, they appeared not to have taken full advantage of the potential of RE:PEAT to enhance cooperation. Because of their focus on physical measures, their awareness of the perspectives of other stakeholders was markedly lower than the corresponding awareness in the other workshops. Furthermore, their overall progress was relatively limited, with other participants contributing significantly less to their overall progress than in the workshops 5–9. Our findings suggest that interventions that stimulate deliberation can prevent individualistic strategies, and instead foster cooperative attitudes. In our research, we achieved this by requiring mandatory deliberation of governmental decisions, which fostered cooperative attitudes in workshops 5–10. Other examples of interventions that can enhance cooperation are scripted instructions (Rummel and Spada, 2005), the incorporation of sequential phases with mandatory group tasks (Hämäläinen, 2011), and scripted collaboration with workshop participants (Papadopoulos et al., 2013) or their virtual counterparts (Hummel et al., 2011).

Our experiment with various application styles also produced some marked results. The participants in the guided workshop spent significantly more time on reflection than the hands-on participants in the other workshops. Consequently, they implemented markedly less actions hour^{-1}, and employed strategies that were less often aimed at influencing the physical system. In addition, in the post-workshop questionnaires, they claimed significantly less often that their understanding of the peatlands had improved. This might suggest that when the ISS application is primarily aimed at increasing the participants' understanding of the social–ecological system, a hands-on approach such as in multi-player serious gaming sessions is preferable to a guided approach such as in touch table sessions. However, we only regarded one workshop with a guided setting, which does not suffice to draw valid conclusions. Further research is needed to examine to what extent the results might have been caused by other factors, such as the limited time that was allocated to the plenary introduction and the debriefing in-between rounds. Furthermore, the impact of the facilitator and the technical assistants was underexposed in our research. The facilitator and technical assistants of workshop 10 had contributed to 5–9 previous workshops, in which they acquired the skills to effectively support an ISS workshop. Arguably, their performance contributed considerably to the added values that were perceived by the participants. Further research might increase our understanding of how ISS facilitators can help improve the support of environmental management.

Context

Van Enst et al. (2014) identify two main contextual factors that define the "structuredness" of policy problems: consensus on relevant norms and values, and certainty about relevant knowledge. On the one hand, unstructured or "wicked" problems are deficient on both accounts. Therefore, they lack a definite solution and exhibit a range of strategical and operational science–policy interaction problems. On the other hand, completely structured problems are characterized by consensus as well as certainty, which makes science–policy interactions relatively straightforward. The context of Dutch peatlands can be seen as a moderately structured policy problem. Regarding knowledge, several Science–Policy Interfaces have increased the certainty of generic knowledge, but some uncertainty still remains, especially regarding site-specific effects from specific stakeholder perspectives. In addition, the workshop participants perceived a trend toward consensus among the stakeholders in Dutch peatlands.

For the workshop participants, the absence of conflicts was an important condition for the high added values they perceived. We therefore believe our results primarily demonstrate how ISS can be of added value for moderately structured environmental management problems. Arguably, ISS might be less effective in unstructured contexts, with less consensus on norms and values and less certainty about relevant knowledge. We suggest that in these contexts, efforts to support environmental management should primarily be aimed at Science–Policy Interfaces which are suited for such contexts, such as processes of participatory knowledge development, boundary organizations, and individual science–policy mediators (Van Enst et al., 2014).

5.5 Conclusions

Our research demonstrated that ISS can improve the support of environmental management. Implementation challenges for collaborative management strategies can be overcome by translating generic scientific knowledge into site-specific effects from multiple stakeholder perspectives and by raising awareness of mutually beneficial strategies. In ISS workshops, all participants explored the physical system dynamics, implementing measures, and the social system dynamics, brokering deals with other stakeholders. As a result, the ISS workshops enhanced cooperation among them and increased their understanding of problems and action perspectives regarding the social–ecological system. Interventions that stimulate deliberation during the ISS workshops were shown to prevent individualistic strategies, and instead foster cooperative attitudes.

The embeddedness of an ISS in preceding Science–Policy Interfaces is an important condition for the ISS's credibility and legitimacy. Important conditions for the salience of the ISS are an abundance of detailed information, realistic visual quality of the user interface, and calculation times that are short enough to keep pace with stakeholder interactions during the decision-making processes. In addition, the general purpose of ISS should reflect capacity building at the individual and group level, striving for a site-specific awareness of the effects of measures and strengthening the resolve of the stakeholders to collectively implement these measures.

We suggest further research on interactive simulation systems should capitalize on the ability of our research approach to yield in-depth understanding of how ISS can improve environmental management. Cross-analyzing questionnaires with logfiles and videos of workshop proceedings can pinpoint how ISS can effectively harness science for complex environmental management tasks. This will help us understand how sustainable management of social–ecological systems can be put into practice.

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Appendix 5A.

<u>Sensitivity analysis regarding the included stakeholder roles</u>

We found only limited changes (Table 5A1–4). However, on several occasions, the limited changes in combination with the smaller group sizes affected the number of groups, yielding significantly different results. Regarding the added value for cooperation, the 4% increase in the average for group 10 resulted in significant differences (workshops 1–4: U=38.0, p=0.018; workshops 5–9: U=44.0, p=0.019). Regarding the argument "more awareness of other people's perspectives", the difference between workshops 1–4 and workshop 10 was no longer significant (p=0.114), because a slightly larger proportion of participants in workshops 1–4 used this argument. A similar effect was found regarding the argument "improved understanding of the peatlands". A slightly larger proportion of participants in workshop 10 who used this argument rendered the difference with the other workshops statistically insignificant (workshops 1–4: p=0.114; workshops 5–9: p=0.054). An opposite effect was found regarding the argument "support of the group process". Because none of the remaining participants in workshop 10 used this argument, the difference between workshops 1–4 and workshop 10 was found to be statistically significant (p=0.030).

Table 5A1. Changes in the results due to the exclusion of the stakeholder roles "collective residents" and "NGO", in terms of the average added values perceived per group of workshop participants for enhancing cooperation and increasing understanding, and the proportion per workshop group that used an argument to explain their perceptions. The scale of added values ranges from -2 (very negative) to 2 (very positive). Changes in the significance of the differences between groups of workshops (p<.05) are asterisked. See Table 5.2 for the results that include all stakeholder roles.

Workshop group	1–4	5–9	10	All
No. of participants	24	26	16	66
Added value for enhancing cooperation	0.0 *	0.0 *	0.2 *	0.0
Added value for increasing understanding	0.0	0.0	-0.1	0.0
More awareness of other people's perspectives	6% *	-12% *	3% *	-2%
Improved understanding of the peatlands	-2% *	7% *	5% *	3%
Support of the group process	-3% *	-10%	-15% *	-9%
Better-informed decisions	-2%	1%	0%	0%
Context of the application	7%	10%	-8%	4%
Usability of RE:PEAT	11%	1%	-16%	1%

Regarding the proportion of time spent on interaction, the averages of workshops 1–4 and workshop 10 became slightly more similar. As a consequence, for the smaller group sizes, the difference between workshops 1–4 and workshop 10 was not statistically significant (U=57.0, p=0.103). Regarding the proportion of workshop participants employing a certain strategy, we found that because the proportion of participants who employed strategies aimed at influencing the system was 9% higher in workshops 5–9 and 12% lower in workshop 10, the difference between these groups was statistically significant (p=0.004). Although the changes regarding the attitudes and the impressions of interaction with other participants were also limited, they affected the number of groups with significantly different results. On the one hand, the average attitude of the participants in workshops 1–4 became slightly less cooperative, which was enough to prevent a significant difference vis-à-vis workshops 5–9 (U=108.0, p=0.128). On the other hand, the average attitude of the participants in workshop 10 became slightly more cooperative, resulting in a significant difference vis-à-vis workshops 5–9 ($U=36.0$, $p=0.038$). The average impression of the interaction was also more cooperative, which resulted in a significant difference vis-à-vis workshops 1–4 ($U=24.0$, $p=0.010$).

Table 5A2. Changes in the results due to the exclusion of the stakeholder roles "collective residents" and "NGO", in terms of the average number of actions and interactions hour-1 per group of workshop participants, and their average proportion of active time spent on exploration, reflection, and interaction. Changes in the significance of the differences between groups of workshops (p<.05) are asterisked. See Table 5.3 for the results that include all stakeholder roles.

Workshop group	1–4	5–9	10	All
No. of participants	24	26	16	66
Actions hour^{-1}	-0.3	0.8	-0.3	0.2
Interactions hour^{-1}	0.4	3.7	4.0	2.6
Time spent on reflection	-1%	-2%	-4%	-2%
Time spent on exploration	0%	-1%	1%	0%
Time spent on interaction	1% *	3%	3% *	2%

Table 5A3. Changes in the results due to the exclusion of the stakeholder roles "collective residents" and "NGO", in terms of the average extent to which groups of workshop participants reached their goals, their average progress score, and the average extent to which their progress was caused by their own actions or the actions of other participants. Note that the significance of the differences between groups of workshops did not change. See Table 5.4 for the results that include all stakeholder roles.

Workshop group	1–4	5–9	10	All
No. of participants	24	26	16	66
Common goal: less soil subsidence	0%	0%	0%	0%
Individual goals	0%	-7%	-4%	-4%
Overall progress score	0%	-4%	-3%	-2%
Progress due to own actions	1%	-1%	-3%	0%
Progress due to actions of others	-1%	-3%	-2%	-2%

Table 5A4. Changes in the results after excluding the stakeholder roles "collective residents" and "NGO", in terms of the proportion per group of workshop participants that employed a strategy, and their average attitude. The scale of attitude ranges from -3 (very uncooperative) to 3 (very cooperative). Changes in the significance of the differences between groups of workshops (p<.05) are asterisked. See Table 5.5 for the results that include all stakeholder roles.

Workshop group	1–4	5–9	10	All
No. of participants	24	26	16	66
Influence the social system	-8%	-6%	9%	-4%
Influence the physical system	-4%	9% *	-12% *	0%
Improve the personal welfare	7%	-3%	15%	5%
Improve the peatlands	6%	3%	0%	3%
No strategy	2%	2%	-12%	-1%
Attitude	-0.1 *	0.0	0.3 *	0.0

Sensitivity analysis regarding the duration of the interactive simulation

We found only limited changes (Tables 5A5–8). However, on four occasions, the limited changes in combination with the smaller group sizes affected the number of groups with significantly different results. First, the difference between workshops 5–9 and workshop 10 regarding the proportion of participants that used the arguments "improved understanding peatlands" was no longer significant (p=0.140). Second, the proportion of workshop participants employing strategies aimed at influencing the physical system differed significantly between workshops 5–9 and workshop 10 (p=0.028). Third, because the reduction of soil subsidence was less in workshops 1–4 and more in workshops 5–9, the difference between these groups was statistically significant (U=40.0, p=0.032). Fourth, the average attitude in workshops 1–4 was more cooperative. As a result, the difference vis-à-vis workshops 5–9 was no longer significant (U=149.0, p=0.342). This effect can be explained by the exclusion of the negative attitudes prevailing in workshops 3 and 4.

Table 5A5. Changes in the results after excluding short workshops, in terms of the average added values perceived per group of workshop participants for enhancing cooperation and increasing understanding, and the proportions of the groups that used an argument to explain their perceptions. The scale of added values ranges from -2 (very negative) to 2 (very positive). Changes in the significance of the differences between groups of workshops (p<.05) are asterisked. See Table 5.2 for the results that include all workshops.

Workshop group	1–4	5–9	10	All
No. of participants	20	25	22	67
Added value for enhancing cooperation	0.0	0.1	0.0	0.0
Added value for increasing understanding	0.0	0.0	0.0	0.0
More awareness of other people's perspectives	-3%	2%	0%	0%
Improved understanding of the peatlands	7%	-7% *	0% *	-2%
Support of the group process	8%	0%	0%	2%
Better-informed decisions	2%	1%	0%	1%
Context of the application	-5%	-9%	0%	-8%
Usability of RE:PEAT	-5%	11%	0%	3%

Table 5A6. Changes in the results after excluding short workshops, in terms of the average number of actions and interactions hour-1 per group of workshop participants, and their average proportion of active time spent on exploration, reflection, and interaction. Note that the significance of the differences between groups of workshops did not change. See Table 5.3 for the results that include all workshops.

Workshop group	1–4	5–9	10	All
No. of participants	20	25	22	67
Actions hour^{-1}	0.7	3.0	0.0	0.4
Interactions hour^{-1}	1.5	1.6	0.0	0.3
Time spent on reflection	-1%	0%	0%	2%
Time spent on exploration	-1%	1%	0%	-1%
Time spent on interaction	1%	-1%	0%	-1%

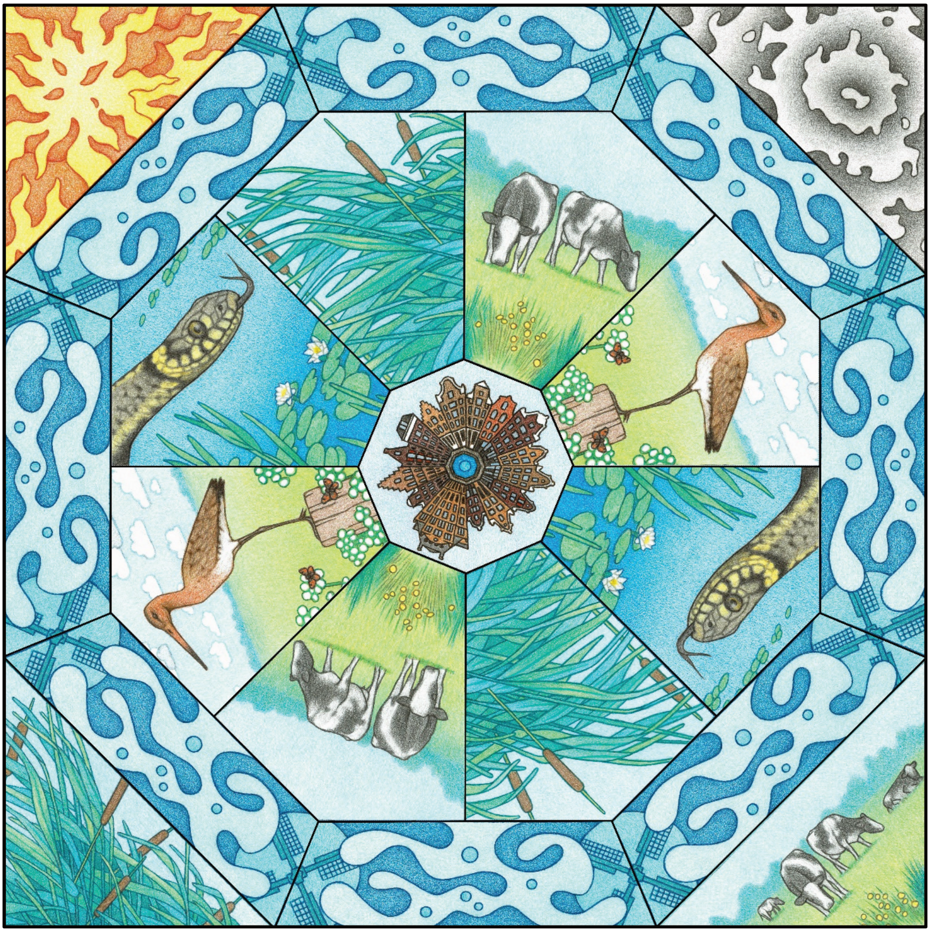
Table 5A7. Changes in the results after excluding short workshops, in terms of the average extent to which groups of workshop participants reached their goals, their average progress score, and the average extent to which their progress was caused by their own actions or the actions of other participants. Note that the significance of the differences between groups of workshops did not change. See Table 5.4 for the results that include all workshops.

Workshop group	1–4	5–9	10	All
No. of participants	20	25	22	67
Common goal: less soil subsidence	-3% *	10% *	0%	5%
Individual goals	10%	2%	0%	4%
Overall progress score	6%	5%	0%	4%
Progress due to own actions	4%	1%	0%	1%
Progress due to actions of others	2%	3%	0%	4%

Table 5A8. Changes in the results after excluding short workshops, in terms of proportions of groups of workshop participants that employed a strategy, and their average attitude. The scale of attitude ranges from -3 (very uncooperative) to 3 (very cooperative). Changes in the significance of the differences between groups of workshops (p<.05) are asterisked. See Table 5.5 for the results that include all workshops.

Workshop group	1–4	5–9	10	All
No. of workshop participants	20	25	22	67
Influence the social system	1%	-3%	0%	-1%
Influence the physical system	3%	13% *	0% *	4%
Improve the personal welfare	10%	6%	0%	5%
Improve the peatlands	9%	10%	0%	7%
No strategy	1%	-6%	0%	-2%
Attitude	0.3 *	0.1 *	0.0	0.2

Chapter 6
Synthesis



To embrace the complexity of social–ecological systems, we need science–policy interfaces that address the socio-economic and the biophysical system dynamics and make the complex information understandable and interactively accessible. The ensemble of the constituent elements of the figure depict the complexity of peatlands. The science–policy interface that is developed in this thesis is especially tailored to handle this complexity.

6.1 Summary of the chapters

A four-step approach towards informed science–policy interactions

This thesis has demonstrated how to design and implement SPIs for the collaborative management of SESs (Fig. 6.1). The heart of the matter consisted of a four-step approach that can help overcome implementation challenges and guide us towards informed science–policy interactions. This section summarizes the lessons that were learned, by presenting the main results and conclusions of the previous chapters.

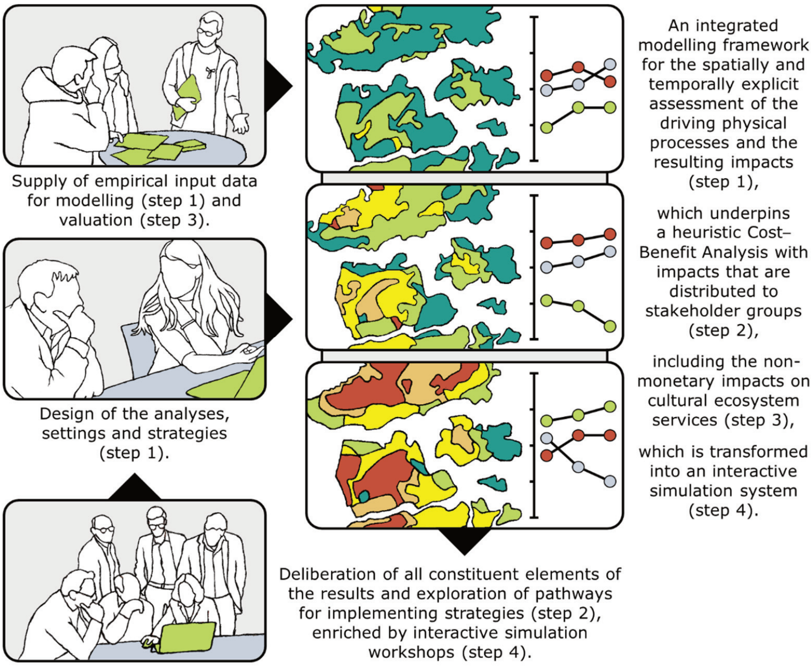


Fig. 6.1. Design of the science–policy interface that is developed in this thesis, consisting of integrated analytical and interactive decision support tools embedded in participatory processes. The triangles indicate the direction in which the features of the interface feed into each other. The steps in which the features are introduced are given in parentheses.

Step 1: An integrated modelling framework

The first step in the development of the SPI (Chapter 2) focused on an integrated modelling framework for the support of the collaborative management of peatlands. The modelling framework was used to assess the long-term impacts of a range of water management strategies in the Dutch peatlands. The results showed that the design of the modelling framework (see section 6.2) made it more useful for the support of collaborative management strategies in peatlands:

- As its core consisted of a GIS-based model that simulated the governing physical processes of the peatlands, abundant and consistent input data were generated for all subsequent impact assessment models that were integrated in the modelling framework.

- As it integrated assessments of all relevant societal impacts, cross-analyses could be made that enriched the understanding of the peatland dynamics, which could be a strong incentive to focus management strategies on long-term impacts rather than on short-term problems.
- As the analysis was temporally and spatially explicit, it improved awareness of the long-term impacts of management strategies.
- As the impacts were distributed to all the stakeholder groups involved, they revealed which stakeholders were disproportionally exposed to the consequences of management strategies, which could make negotiation processes on goals, means, and future pathways more transparent.

It is believed that sustainable management strategies of peatlands should address the interests of all stakeholders, should slow down peat loss, and should ensure the required long-term settings for the slower subsidence rates (Joosten and Clarke, 2002; Den Uyl and Wassen, 2013). Chapter 2 demonstrated the necessity of integrated analytical support tools that acknowledge this multi-dimensional perspective. The integrated modelling framework transcended previous support tools which only regarded a limited number of societal impacts (Schothorst, 1977; Wösten et al., 1997; Hooijer et al., 2010; Verhoeven and Setter, 2010; Querner et al., 2012). As a result, it enhanced our understanding of the complex peatland dynamics, which is regarded as a necessity to design sustainable management strategies for complex SESs (Ostrom, 2007).

The added value of the modelling framework clearly resulted from its use by the regional water authority of the research area. They used the findings as input for a CBA, which provided a clear incentive for more sustainable management practices. In fact, they improved a policy that was revealed to have unwelcome long-term socio-economic impacts. This example confirmed the observations by Runhaar (2016), who pointed out that the usefulness of solely analytical integration tools is usually modest, precisely because they fail to provide insight into socio-economic consequences or fail to deal with controversies and conflicting interests. The integrated modelling framework can have a more profound impact, because it addresses these issues better, especially in combination with CBA. This insight was used to direct the next step in the development of the SPI.

<u>Step 2: Cost–Benefit Analysis as a heuristic aid</u>

The second step in the development of the SPI (Chapter 3) expanded on the results of the first step. A heuristic CBA, underpinned by the modelling framework, exposed that a top-down approach to raise water levels and achieve a transition in land use was not viable because gains and losses would be unequally distributed. The participating stakeholders discussed the results at length, focusing on the management problem and propagating the need for collaborative adaptations and context-specific follow-up processes. The results revealed the added value of the analytical modelling framework in combination with a heuristic CBA for the support of the collaborative management of SESs because of the following:

- The clear participatory design of the heuristic CBA processes successfully enabled the involvement of stakeholders.
- The comprehensive presentation of the constituent elements of the results from multiple stakeholder perspectives revealed impacts of management strategies that were not clear beforehand.

- The abundant opportunities to discuss the results across multiple perspectives increased the stakeholders' understanding of the complexities of SESs and their ability to explore the potentialities of these systems.

The results demonstrated ways to overcome process issues that have been reported to inhibit the usefulness of CBA for collaborative processes (Beukers et al., 2012; Mouter et al, 2013). This allowed the CBA to deliver on its potential for the support of collaborative processes, as has been perceived by many scholars (De Jong and Geerlings, 2003; Turner, 2007; Browne and Ryan, 2011; Beria et al., 2012). However, several limitations remained that required further attention. First, the participants of the case study stressed the need for an additional evaluative endpoint for non-monetary effects such as cultural heritage. Their opinions reflected the proposition of some scholars that a more holistic scope of CBA is needed, including less tangible and indirect impacts (Browne and Ryan, 2011; Beria et al., 2012). Second, the transparency of the approach and its ability to handle uncertainty was essential for the interpretation of the results. This feedback was used to direct the third and fourth steps of the development of the SPI.

<u>Step 3: Participatory non-monetary valuations</u>

The third step in the development of the SPI (Chapter 4) further added to the insights from the first and second steps. The results revealed how a heuristic CBA underpinned by quantitative modelling could provide a platform to integrate non-monetary valuations in the collaborative management of SESs, enriching the decision-making process. The added value of the approach for the support of the collaborative management of SESs is the following:

- It successfully engaged key stakeholder groups by using a participatorily designed valuation strategy, a well-advertised internet survey, and deliberative workshops.
- It presented spatially explicit valuation maps of cultural ecosystem services from multiple stakeholder perspectives, revealing the spatial consequences of preferences that were not clear beforehand.
- It combined monetary assessments with non-monetary valuations, supported by a deliberative mapping tool, which instigated a process of value deliberation between stakeholder groups and allowed for negotiation across multiple evaluative endpoints.

The results demonstrated how the participatory mapping of robust, transparent, and stakeholder-relevant cultural ecosystem services is key to integrating monetary and non-monetary assessments. Most previous research on mapping practices reflected a regional, national, or continental scale, frequently with a limited number of cultural ecosystem services (Hernández-Morcillo et al., 2013). The research in this thesis reflected a local scale that better fitted the collaborative management process. Moreover, it considered a wide range of cultural ecosystem services and related elements of planning options without an ecosystem origin, which effectively captured lay perceptions of the SES. This enhanced participatory mapping approach instigated a discussion among the stakeholders on how differences in transcendental values had shaped their valuation of the research area. If these value deliberations are embedded in a multi-loop social learning process, they may result in the formation of shared cultural values beyond the aggregated utilities of individuals, which may contribute to reaching consensus on collaborative management practices. The approach presented in this thesis can effectively support this task, provided that its interactive features are further enhanced. The subsequent, final step in the development of the SPI was aimed at meeting this challenge.

Step 4: Interactive simulations

The fourth step in the development of the SPI (Chapter 5) demonstrated how the SPI created in steps 1–3 could be transformed into an ISS. As the ISS was applied in ten workshops attended by real-world stakeholders and was evaluated using logfiles, videos, and post-workshop questionnaires, an in-depth perspective was derived that revealed the added value of ISSs to overcome implementation challenges for collaborative management strategies. In the workshops, all participants explored not only the physical system dynamics by implementing measures, but also the social system dynamics by brokering deals with other stakeholders. In particular, the participants collaboratively designed adaptive drainage strategies that could slow down soil subsidence. Although the generic effects of these adaptive drainage strategies had been known for approximately a decade (Querner et al., 2012), they had never been implemented on a large scale. As the ISS translated generic scientific knowledge into site-specific effects from multiple stakeholder perspectives, it created an operational fit between knowledge supply and demand. In addition, because the ISS supported informed negotiations, it also raised awareness of mutually beneficial strategies. As a result, cooperation among the workshop participants was enhanced and their understanding of problems and action perspectives regarding the SES increased.

The research showed that the general purpose of the ISS should reflect capacity building at the individual and group level, striving for a site-specific awareness of the effects of measures and strengthening the resolve of stakeholders to collectively implement these measures. In addition, the cross-analyses from multiple methodological viewpoints provided further insight into how the ISS can effectively harness science for the collaborative management of SESs. Regarding the system design, the research revealed how the embedding in preceding SPIs enhanced the SPI's credibility and legitimacy, and how its salience was strengthened by abundant detailed information, realistic visual quality, and calculation times that were short enough to keep pace with stakeholder interactions during the decision-making processes. Regarding the workshop setting in which the system was applied, the research revealed that interventions must be incorporated that prevent individualistic strategies but that foster cooperative attitudes instead.

6.2 General discussion and conclusions

This thesis started with the observation that a clear understanding of how to design and implement SPIs for the collaborative management of SESs is a persistent knowledge gap in the scientific literature. The research in Chapters 2–5 aimed to fill this knowledge gap by designing and implementing an SPI that can advance the collaborative management of SESs. This section discusses to what extent that aim was reached. First, the design principles for the development of the SPI and the conditions for its successful implementations are discussed. Second, the context of the research is reflected upon. Third, general conclusions are drawn about the research.

Towards informed science–policy interactions

In general, all SPIs aim for informed science–policy interactions that produce useful knowledge. Van de Riet (2003) argues that the production of useful knowledge in a multi-actor policy setting requires that the analyses must (a) be perceived as trustworthy by the participating stakeholders, (b) bridge interests by fully comprehending and considering the interests of all stakeholders and adopting a broad scope, and (c) take a multi-perspective research focus covering all features that the stakeholders deem relevant, considering multiple views on policy issues, and providing insight into the distribution of gains and losses that result from management strategies. She suggests a variety of approaches to reach these goals, such as involving trusted analysts, establishing a system of checks and balances, letting stakeholders participate in the analyses, and enhancing the accessibility and understandability of the results. Regarding the suggested broader scope, Kørnøv and Thissen (2000) propose that impact assessments should also consider the interrelations between actors, which could help identify workable management strategies, and that they should include values in addition to facts, which will help clarify diverging viewpoints and build bridges between them.

All these suggestions were put into practice during the research of this thesis. The decision support tools elaborated on previously produced knowledge that was already known and trusted by many stakeholders, such as empirical soil subsidence equations (Van den Akker et al., 2008) and Dutch CBA guidelines (Eijgenraam et al., 2000; Faber and Mulder, 2012; Romijn and Renes, 2013). Moreover, several experts who had previously produced this knowledge were also involved in the design of the decision support tools that were used during the research of this thesis. Much effort was put into the deliberation of the results, supported by easily understandable graphical designs (Fig. 2.3 and Fig. 4.5), a web-based tool for evaluating the quality of the research (step 2), and a deliberative mapping tool (step 3). The stakeholders actively contributed to the analyses by supplying input data and co-designing the impact assessment and the CBA. Their participation markedly broadened the scope of the impact assessments. Until now, impact assessments in peatlands have mainly focused on the physical process of soil subsidence and a limited number of impacts (Schothorst, 1977; Wösten et al., 1997; Hooijer et al., 2010; Verhoeven and Setter, 2010; Querner et al., 2012; Brouns et al., 2015). By integrating additional assessments that were co-designed by the participating stakeholders, the entire range of their interests was more effectively reflected, and this enhanced the salience and legitimacy of the impact assessment. In addition, participatory evaluation of cultural ecosystem services in combination with monetary assessments was shown to instigate a process of value deliberation between stakeholder groups, which allowed for negotiation across multiple evaluative endpoints. Therefore, a participatorily designed, broad scope of the impact assessments is advised to enhance the usefulness of SPIs.

Further design principles to help realize informed science–policy interactions can be derived from Cash et al. (2003), who suggest effective boundary management has at least three organizational features that facilitate communication, translation and mediation across boundaries: (1) dual accountability to scientists and practitioners, (2) specialized roles for managing the boundary between science and practice, and (3) the use of boundary objects. These suggestions were also implemented in the research of this thesis.

To achieve dual accountability to scientists and practitioners, a joint effort of scientists and practitioners was required, which simultaneously enhanced the societal relevance and the scientific credibility of the research. A key requirement for the successful implementation was the integration of scientific and lay perceptions of the SES. On several occasions, a rigid application of evaluation doctrines would not have captured all the concerns that were relevant for stakeholders, such as the indirect effects within the agricultural supply chain (step 2) or the elements of the planning options that negatively affected cultural ecosystem services (step 3). However, because the heuristic evaluation of the results allowed for multiple evaluative endpoints, a lenient stance towards the evaluation doctrines was permissible and laypeople's concerns could be addressed without reducing the credibility of the results.

The suggested specialized roles for managing the boundary between science and policy were implemented in several ways. Experts were called upon to assist in communicating and translating the knowledge, such as landscape architects, who translated the general framework for valuing ecosystem services into images that were meaningful and easily understandable from a lay perspective (step 3), and facilitators and technical assistants, who led the ISS workshops (step 4). In addition, several participating stakeholders in the case study spontaneously adopted such specialized roles, for example by arranging meetings with board members of other organizations to discuss the results and reconcile conflicting policies of their organizations (step 2) or by engaging colleagues in a collaborative project aimed at implementing the lessons that had been learned (step 4). These initiatives reflect a design principle for joint knowledge production that is perceived by Hegger and Dieperink (2014b), in the sense that these "boundary-spanning participants" transcended the stereotypical view of their role, mediating between the worlds of science and policy. The importance of engaging boundary-spanning participants is also illustrated by Runhaar and Van Nieuwaal (2010), who explain an improvement in the SPI of the Dutch Wadden Sea by better alignment of research and decision-making processes, which was achieved by a boundary-spanning mediator.

The implication of the research of this thesis for the implementation of SPIs is that the boundary management functions of communication, translation, and mediation across boundaries do not necessarily need to be organized in advance. Instead, these functions can be assigned to the participants of the collaborative management process, provided that the SPI supports capacity building for translating generic scientific knowledge into site-specific effects from multiple stakeholder perspectives and increases awareness of mutually beneficial strategies. In other words, the SPI must instigate a process of social learning.

Reed et al. (2010) point out that social learning requires more than just the participation of stakeholders in the decision-making process. Learning also requires that a change in understanding has occurred, either at a superficial level, regarding the consequences of actions and behavior, of at a deeper level, reflecting a change in assumptions, norms, and values. The research of this thesis instigated changes in understanding on several occasions, which all relate to the use of boundary objects. For example, the DialogueMaps tool that was used in step 3 was shown to support a discussion among the stakeholders on how differences in transcendental values shaped their valuation of the research area, which enabled negotiation across multiple evaluative endpoints. Another boundary object that resulted in social learning was the RE:PEAT tool (step 4). The participants of the workshops in which RE:PEAT was applied unequivocally reported a high added value of the tool for increasing their understanding (Table 5.2). It is noteworthy that later, several workshop participants initiated a collaborative process to implement the adaptive drainage strategy designed during the workshops in the same peatland area that they had explored with RE:PEAT. Moreover, they intend to continue using RE:PEAT as a boundary object, to support collaborative management decisions in the years to come.

The usefulness of RE:PEAT as a boundary object can be attributed to the integration of analytical and interactive decision support tools into an ISS. Hands-on participation in interactive simulation workshops was shown to increase the capacity of the stakeholders to understand the complexities of the SES and their ability to explore its potentialities. As RE:PEAT created a site-specific awareness of the effects of measures and strengthened the resolve of the stakeholders to collectively implement these measures, a process of social learning was instigated. It must be noted that an ISS does not automatically result in knowledge that is perceived as credible, legitimate and salient. For example, White et al. (2010) report how the stakeholders who evaluated the ISS WaterSim were at first rather skeptical about its usefulness, because they had been insufficiently involved in its design and the scope of the impact assessments was too narrow. This compelled the developers to request input from a broader range of stakeholders, to ensure that their knowledge demands were adequately addressed.

The importance of including a wide range of stakeholders is remarked upon by several other scholars. For example, Reed et al. (2010) point out that social learning requires that a change in understanding transcends the individual level and becomes situated in wider social units or Communities of Practice, through processes of social interaction between the participating actors. Hegger and Dieperink (2014b) argue that the inclusion of a broad coalition of scientists and practitioners is key to enhancing the effectiveness of processes of participatory knowledge development. In particular, they advocate the creation of "a protected space for knowledge development while establishing connections with ongoing policy processes". This design principle is also acknowledged by Boezeman et al. (2013), who use it to explain the success of the Dutch Delta Committee's functioning as a boundary organization.

The research of this thesis duly considered this design principle, with many scientists contributing to the design of the SPI, 240 stakeholders discussing the results of the heuristic CBA (step 2), 295 stakeholders contributing to the evaluation of cultural ecosystem services (step 3), and 89 stakeholders participating in the RE:PEAT workshops (step 4). Remarkably, the broad participation resulted in a Community of Practice that focused on enhancing the usability of RE:PEAT as a widely acknowledged boundary object within wider social units, for example by organizing free training courses and jointly expanding the ability of the ISS to meet additional knowledge demands. This example illustrates that if an SPI for the collaborative management of an SES supports social learning processes, the SPI is more likely to advance the collaborate management of the SES.

Arguably, social learning processes require more than boundary objects. Due attention must be paid to the design of effective participatory processes in which the boundary objects are used. Van den Hove (2007) points out that an SPI must be seen as a value-laden social process. Therefore, she emphasizes the need for transparent rules for continuous, dynamic exchange and co-evolution of knowledge, including ample reflection on research priorities. She also argues that SPIs on the one hand allow for genuine interdisciplinary interactions between social and natural sciences, and on the other hand exploit the potential of social sciences for translating and mediating between different groups of actors. Her suggestions resemble the conditions regarding discourses and rules that Hegger and Dieperink (2014a) expect to enhance the salience, credibility, and legitimacy of joint knowledge production.

Regarding the facilitation of dynamic and interdisciplinary processes, the research of this thesis has clearly demonstrated the merits of heuristic evaluations for the impact assessments. Traditionally, approaches for project appraisals such as CBAs are used as a decision-rule to identify the optimal alternative, which conflicts with the iterative manner of consensus building in collaborative management processes (Turner, 2007; Beukers et al., 2012; Mouter et al., 2013). Therefore, many scholars suggest that CBAs should refrain from passing final verdicts, but instead be used as a method for collecting, organizing, and discussing information relevant to collaborative management processes (De Jong and Geerlings, 2003; Turner, 2007; Browne and Ryan, 2011; Robinson and Hammitt, 2011; Beria et al., 2012). By presenting and discussing all the constituent elements of the assessments, it was shown that CBAs could indeed be used heuristically. As a consequence, negotiation processes could be enriched by cross-analyses of spatially and temporally explicit assessments from multiple stakeholder perspectives, which reveal impacts of management strategies that were not explicit beforehand.

Context matters

The Dutch peatlands serve as an illustrative context for all the research that is presented in this thesis. Consequently, before overall conclusions can be drawn, some reflections must be made on the generalizability of the results to other SESs.

Environmental governance in the Dutch peatlands is multi-faceted, with elements of centralized and interactive governance modes (Chapter 1). To illustrate which modes can be supported by the SPI developed in this thesis, the conceptual framework of Driessen et al. (2012) to differentiate between modes of environmental governance can be applied. Their framework compares five archetypical arrangements between state, market, and civil society, which include centralized and interactive governance modes. The comparison is based on a range of features of actors, institutions, and policy content. Regarding actor features, interactive governance reflects power based on agreement, trust, and knowledge, whereas in centralized governance power is based on coercion, authority, and democratic representation. Regarding institutional features, interactive governance reflects mechanisms for social interaction such as social learning, deliberations, and negotiations, whereas centralized governance entails top-down, command-and-control mechanisms for social interaction. Regarding policy content features, interactive governance emphasizes integrated and time-and-place-specific knowledge, whereas in centralized governance, generic expert knowledge is more prominent. Regarding all these features, the SPI developed in this thesis is specifically suited for the interactive governance mode, but poorly equipped for the centralized governance mode. Therefore, the results of this thesis are primarily relevant for SESs with an interactive governance mode.

It is noteworthy that the interactive aspects of the governance of the Dutch peatlands entails a distinct feature that, at first sight, might be of consequence for the generalizability of the results, namely the long tradition with consensus-based policy making, i.e., the Polder model. However, after closer scrutiny, there appears to be no reason to romanticize this notion. Soens (2006) points out that in medieval water management, asymmetrical power relations often favored the political and economic elite, even though formal attention was paid to participation and broad consultation. Therefore, the interactive mode of governance in the Dutch peatlands appears comparable to similar modes in other SESs.

Although the SPI developed in this thesis is specifically aimed at interactive governance, its effectiveness arguably depends on the preceding SPIs that were created to enhance decision-making in the Dutch peatlands. In this regard, it is noteworthy that in the Dutch peatlands, various processes of participatory knowledge development have been instigated and a boundary organization for innovative peatland management has been created. This has increased the generic knowledge, improved the understanding of the impacts of innovative adaptations, and provided guidance for regional adaptation strategies (Van Brouwershaven and Lokker, 2010; Brouns et al., 2015; Driessen et al., 2015. The SPI developed in this thesis capitalized on the results of these preceding SPIs, aiming to improve operational science–policy interactions. As a consequence, the results of this thesis are generalizable to other SESs where the improvement of operational, site-specific science–policy interactions is the prime concern. In SESs where the generic knowledge and/or the impacts of adaptations are uncertain, generalizability may be somewhat less straightforward. In these contexts, it is advisable to direct the main efforts towards the creation of SPIs that are well-suited to improve the generic knowledge and/or the impacts of adaptations. For example, Van Enst (2018) suggests that in these contexts, the most effective SPIs are boundary organizations from outside the scientific and policy arenas, individual boundary-spanning mediators, and processes of participatory knowledge development aimed at creating boundary objects. In a subsequent phase of the collaborative management process, these SPIs can be succeeded by the SPI developed in this thesis.

Conclusions

This thesis has demonstrated how to design and implement an SPI for the collaborative management of SESs that consist of integrated analytical and interactive decision support tools embedded in participatory processes. The design of the SPI considered several general principles that were derived from scientific literature (Table 6.1). The SPI was implemented in the Dutch peatlands. Despite several previous SPIs, the collaborative management in this SES was hampered by operational misfits between the supply and demand of knowledge. The research of this thesis demonstrated how these implementation challenges could be overcome, and the collaborative management of the SES could be advanced. The successful implementation of the SPI enabled informed science–policy interactions that enriched the management of the SES.

The conditions for the successful implementation of the SPI (Table 6.1) reflect increased stakeholder participation in the assessments and an improved suitability of the decision support tools for collaborative management processes. The assessments were participatorily designed and had a broad scope which adequately reflected the stakeholders' interests and values. All the constituent elements of the assessments were heuristically evaluated in deliberative workshops. As heuristic evaluation accepts multiple evaluative endpoints, a lenient stance towards evaluation doctrines was permissible and laypeople's concerns could be addressed without reducing the credibility of the results. In addition, special emphasis was placed on enhancing the scientific knowledge from a lay perspective, which strengthened the science–policy interactions. The deliberative workshops were enriched by RE:PEAT, a boundary object which combined analytical and interactive decision support tools. The workshops improved the stakeholders' understanding of the complexities of the SES and their awareness of options to advance its collaborative management. This enabled the stakeholders to play an active role in the mediation between the worlds of science and policy. In addition, they created a Community of Practice that focused on enhancing the usability of RE:PEAT as an acknowledged boundary object within wider social units.

The SPI was developed to advance collaborative management processes. Consequently, it is primarily relevant for SESs with interactive governance modes, but poorly equipped for SESs with centralized governance modes. The SPI is primarily suited to improve operational, site-specific science–policy interactions. Its effectiveness depends on preceding SPIs which have enhanced the generic knowledge of the SES, such as boundary organizations and processes of participatory knowledge development.

Table 6.1. Design principles for science–policy interfaces for the collaborative management of social–ecological systems, and the conditions that enhance their successful implementation.

Design principles	Conditions for successful implementation
Analyses that are trustworthy, bridge interests and take a multi-perspective focus By acknowledging the requirements that a multi-actor setting imposes on the analyses, the usefulness of the knowledge is enhanced (Kørnøv and Thissen, 2000; Van de Riet, 2003).	*A participatorily designed, broad scope of the impact assessments* By integrating assessments and valuations that were co-designed by the participating stakeholders and that adequately reflect their interests and values, the salience and legitimacy of the impact assessment can be enhanced.
Dual accountability to scientists and practitioners By addressing the interests of both scientists and practitioners, the salience, credibility, and legitimacy of the knowledge is enhanced, which results in more effective links between assessment and decision-making (Cash et al., 2003).	*The integration of scientific and lay perceptions of the social–ecological system* By allowing multiple evaluative endpoints, a lenient stance towards evaluation doctrines is permissible and laypeople's concerns can be addressed without reducing the credibility of the results.
Specialized roles for managing the boundary between science and policy By investing in communication, translation, and mediation, the salience, credibility, and legitimacy of the knowledge is more effectively balanced (Cash et al., 2003; McNie, 2007; Van den Hove, 2007).	*The engagement of experts who can assist in communicating and translating the knowledge* By enhancing communication and translation of the scientific knowledge, the knowledge becomes more easily understandable from a lay perspective, which strengthens the science–policy interactions.
Engagement of boundary-spanning participants By engaging participants who transcend the stereotypical view of their role, mediation between the worlds of science and policy is enhanced (Runhaar and Van Nieuwaal, 2010; Hegger and Dieperink, 2014b).	*The assignment of boundary management functions to the participants of the management process* By supporting capacity building and social learning, stakeholders are enabled to play an active role in boundary management, which enhances its effectiveness.
Use of boundary objects By providing a platform to exchange and co-create knowledge, a shared focus is created with room for multiple interpretations, which bridges differences in goals and languages and enhances the salience, credibility, and legitimacy of the knowledge (Cash et al., 2003; White et al., 2010).	*The integration of analytical and interactive decision support tools* By enabling hands-on participation of stakeholders in interactive simulations, a boundary object is created which enhances both the capacity of the stakeholders to understand the complexities of the social–ecological system and their awareness of options to advance its collaborative management.
Inclusion of a broad coalition of scientists and practitioners By creating a protected space for knowledge development while establishing connections with ongoing policy processes, credible knowledge can be produced which is also accepted as legitimate and salient by practitioners (Reed et al., 2010; Boezeman et al., 2013; Hegger and Dieperink, 2014b).	*The creation of a broad Community of Practice that enhances the usefulness of a boundary object* By engaging a broad group of scientists and practitioners, a boundary object becomes acknowledged in wider social units, which enhances its ability to support social learning processes and to advance the collaborative management of the social–ecological system.
Facilitation of dynamic and interdisciplinary processes By enabling dynamic, interdisciplinary exchange and production of knowledge, with transparent rules and discourses, the salience and legitimacy of the knowledge is enhanced (Van den Hove, 2007; Hegger and Dieperink, 2014a).	*Heuristic evaluations of the impact assessments* By presenting and discussing all the constituent elements of the assessments, negotiation processes are enriched by cross-analyses of spatially and temporally explicit assessments from multiple stakeholder perspectives, which reveal the impacts of management strategies that were not explicit before.

6.3 Perspectives

Putting science–policy interfaces into practice

Traditionally, the final section of a thesis reflects on the perspectives for further scientific research and for practical application of the results. This thesis does not make that distinction, because its aim reflected the practical application of scientific research, i.e., a better understanding of how to implement the scientifically established design principles of SPIs in real-world collaborative management processes. Building on this, the perspectives for further scientific research and for practical applications of the results are aligned, which is why they will be discussed in conjunction with each other.

This thesis presented a four-step approach to designing and implementing SPIs for the collaborative management of SESs (Fig. 6.1). Steps 1–3 are key, whereas the fourth step, the creation of an ISS, is optional. A capable mediator will be able to create an effective SPI based on steps 1–3 alone. Moreover, creating an ISS might not be feasible in all circumstances. For example, the creation of the RE:PEAT template took two years and required an investment of more than €250,000. However, whenever feasible, it is strongly advised to include step four. Many stakeholders who participated in the multi–criteria discussion of the cost and benefits of soil subsidence (step 2) requested the possibility of using the analytical tools interactively. Those who participated in the workshops in which the ISS was applied unequivocally perceived a high added value because their understanding of the peatland dynamics and of recognizing opportunities to broker deals had been greatly improved. One of them even remarked at the debriefing of a workshop that he had come to realize that the key to advancing the collaborative management of the Dutch peatlands is the awareness that everything one does affects the other stakeholders and that cooperation is the only way to achieve something. His remark resembled a famous Johan Cruijff quote: "You'll only see it when you understand it." Ultimately, such an enlightened state of mind is exactly what informed science–policy interactions are meant to accomplish.

Besides a high added value for implementing SPIs for the collaborative management of SESs, ISS also offers some interesting options for further development, for both science and practice. Currently, an active Community of Practice is expanding the features of the Tygron Geodesign Platform, which includes the RE:PEAT template. They intend to use the same tool for (a) raising awareness of collaborative management strategies towards more sustainable use of the peatlands, and (b) subsequent detailed impact assessments that can guide the implementation of these strategies. The additional features also enable integrated impact assessments of strategies aimed at adaptations to climate change and improvement of water quality. In particular, integration with the PCDitch model (Janse and Van Puijenbroek, 1998) will enable highly spatially detailed water quality assessment which can be directly linked to the social–ecological dynamics of the peatlands. Such transdisciplinary research requirements are believed to be necessary for the next generation of social–ecological decision support tools (Mooij et al., 2019). It would be interesting to further explore the added value of the Tygron Geodesign Platform for improving ecological management practices.

Another interesting development is the longitudinal use of RE:PEAT as a boundary object. Currently, several collaborative management processes in the Dutch peatlands are using the RE:PEAT tool to underpin novel applications of field drains. The impact of the field drains on water management, farm management, greenhouse gas emissions, and biodiversity are somewhat uncertain. Therefore, joint knowledge production processes are instigated to explore how the field drains can contribute to more sustainable peatland management. It would be interesting to explore what is required for RE:PEAT to maintain its usefulness as a boundary object for these processes. In other words, it would be interesting to explore how the RE:PEAT tool can adapt to the changing knowledge demands of the collaborative management processes, and to analyze to what extent it is used to underpin collaborative management decisions.

The developments regarding ISSs do not only reflect software and management processes, they also present novel challenges for the individuals who apply the software to support the management processes. ISSs will increasingly allow stakeholders to take on the task of assessing impacts, replacing the traditional hydrological or ecological experts. This will generate a bidirectional flow of knowledge throughout a resource management process. An ISS will ensure that the stakeholders receive knowledge that is supplied by experts who have incorporated it into the ISS. In response, the experts will receive additional knowledge demands that are supplied by the stakeholders who operate the ISS. The experts' new task is to respond to these additional knowledge demands from a transdisciplinary knowledge base, taking on the challenge of bridging and integrating the diverging demands. This is no small thing. In fact, in the closing remarks of their influential paper, Cash et al. (2003) even observe that such knowledge systems call for "a radical new contract" to harness science, encompassing not just individual projects, but entire professional careers. It would be interesting to explore how the next generation of experts will meet this challenge, how they will apply ISSs, and what their contribution to more sustainable management of our natural resources will be.

<u>The future of the Dutch peatlands</u>

The Dutch peatlands are the illustrative context for the research that is presented in this thesis. Therefore, it seems suitable that the closing words of this concluding chapter should address this SES in particular. What better starting point for this contemplation than the question that has been asked repeatedly by many scholars (Joosten and Clarke, 2002; Bragg and Lindsay, 2003; Den Uyl and Wassen, 2012): how can a wiser, sustainable use of peatlands be achieved? This thesis proposes that the answer to this question reflects an integrated, collaboratively designed pathway with a fair distribution of costs, benefits, and ecosystem services, which enhances the resilience of the peatlands. However, this answer is rather generic, whereas one of the lessons that can be learned from this thesis is that collaborative peatland management always requires a time-and-place-specific approach.

A recent process of joint knowledge production regarding climate adaptation in peatlands suggests a range of time-and-place-specific adaptation strategies, ranging from more sustainable dairy farming to a "biobased economy" in swamp-like conditions (Kwakernaak et al., 2014; Driessen et al., 2015). Local projects that are aimed at collaborative implementation of such time-and-place-specific adaptation strategies are becoming increasingly common. An example is the application of field drains that reduce soil subsidence, in which the management costs are shared collectively by the participating stakeholder groups (Chapter 3). The SPI developed in this thesis is especially suited to support these first steps towards a more sustainable use of peatlands.

Once the first steps have been taken on the pathway to sustainability, the remaining challenge is to respond adequately to the changes that await us. As Joosten and Clarke (2002) point out, uses that appear wise in the short term may be modified in time, turning out to be unwise in the long-term. Therefore, Den Uyl and Wassen (2012) suggest dealing very cautiously with possible short-term sacrifices. For instance, the National Climate Agreement of the Netherlands aims for a 49% reduction in greenhouse gas emissions by 2030 and a 98% reduction by 2050. This could be achieved by a large-scale transition from dairy farming to paludiculture or to swamps, together with a raising of the water levels. However, such transitions would also obliterate the cultural ecosystem services of the Dutch agricultural landscape (Chapter 4), the breeding habitat quality of meadow birds (Van Dijk et al., 2014), and dairy production. In this regard, it must be noted that the Common Agricultural Policy of the European Union was implemented as a reaction to the devastation and famine during World War II. On the brink of the third decade of the twenty-first century, we may ask ourselves whether we can be certain that history will not repeat itself. Arguably, a more prudent strategy would be to not completely disregard the importance of agricultural production just yet, and to also consider adaptation strategies which deal more cautiously with short-term sacrifices.

If we consider an even longer timeframe, the uncertainty only increases. It is difficult to predict how the Rhine-Meuse delta will evolve. Will the dynamics of sea level rise and soil subsidence lead to a transgression of the shoreline and the inundation of vast expanses of peat deposits, similar to what happened between 2,500 and 500 years ago (compare Fig. 1.1A and 1.1B)? Or will urbanization claim even more of the remaining rural areas than in the current situation (Fig 1.1C)? In short, it is difficult to predict which goals will become most urgent in the decades to come.

Although it clearly remains uncertain what can be considered a wise use in the future, the way to achieve this wise use – whatever it may be – is clearer. Joosten and Clarke (2002) recommend a pluralist stance as the best way to deal with complex situations like this. This recommendation is reflected by the SPI developed in this thesis. Enhancing the resilience of the Dutch peatlands implies time-and-place-specific trade-offs between restoring biodiversity, decreasing soil subsidence, reducing greenhouse gas emissions, and maintaining agricultural production. In locations that are biophysically or economically least suited to dairy farming, a transition towards swamps or paludiculture seems sensible, whereas in locations that are well-suited to dairy farming, a transition towards more sustainable agricultural practices seems a viable alternative. Such time-and-place-specific trade-offs will ensure that the peatlands maintain the ability to respond to the challenges that await us in the decades to come. The SPI developed in this thesis is specifically tailored to support the interactive negotiation processes with multiple evaluative endpoints that are necessary to deal with complex challenges like this. This is particularly true if the SPI is used as a longitudinal boundary object, with iterative adaptations to the changing knowledge demands of the collaborative management processes.

This prospect might appear rather optimistic. However, the question remains whether the envisioned gradual pathway will foster more sustainable management strategies fast enough for a timely restoration of peatland biodiversity and a significant reduction of soil subsidence and greenhouse gas emissions. Some people may argue that faster, transformative rather than incremental changes are needed, and therefore, more compelling action is required. The Dutch National Climate Agreement may be instrumental in bringing about a faster pace of change. For example, if the national government would subsidize the environmental service of reducing greenhouse gas emissions from peatlands, this would markedly enhance the feasibility of implementing more sustainable management strategies for the collective local stakeholders.

In addition, the successful implementation of accelerated adaptation pathways requires a conscious choice regarding science–policy interactions. In situations such as the Dutch peatlands, in which there is not only consensus regarding the problem but also some uncertainty regarding knowledge and/or future system dynamics, science–policy interactions can either focus on generating additional knowledge, or on a conscious, agreed-upon approach to taking risks (Kørnøv and Thissen, 2000). If we truly want to pursue accelerated adaptation pathways, we must embrace both approaches to science–policy interactions. We must implement adaptations and agree upon a decision-making process that enables a rapid response to any unforeseen effects of the adaptations, instead of postponing their implementation until science provides the definitive answers to all resource management questions, particularly as it is unlikely that such an ideal situation will ever be realized.

Fortunately, the SPI developed in this research is specifically suited to support the implementation of collaborative management approaches. In addition, the RE:PEAT template (Chapter 5) is available for all collaborative management processes in peatlands around the world, enabling support in these processes for anyone who desires it. Thus, we stand a better chance of reaching the informed science–policy interactions that are needed for the sustainable use of our natural resources. To emphasize this optimistic prospect, contrary to most research which ends with suggestions for additional research, this thesis ends with the suggestion to simply RE:PEAT it.

References

The medieval water systems and allotment patterns of the Dutch peatlands still exist and are acclaimed as valuable Dutch cultural heritage. The left-hand side of the figure depicts some of the characteristic cultural heritage features of the Dutch peatlands. The right-hand side of the figure depicts a rope yard, which in previous centuries were quite common in the Dutch peatlands, using the hemp that was grown in the peatlands to manufacture ropes for sailing ships. The characteristic Dutch clogs in the center of the figure refer to the man-made, agricultural identity of the Dutch peatlands.

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Summary

Traditionally, the Dutch peatlands were renowned for their species-rich hayfields and ditches. However, the biodiversity has markedly decreased over the last decades. One of the challenges for the next decades is to arrest and reverse this trend. The figure shows several characteristic species of the Dutch peatlands. Yellow iris (left-hand side) is still quite common along well-managed watercourses. Snake's head (center) has become very rare, because it requires very shallow groundwater tables. Marsh-marigold (right-hand side) can still be found along well-managed watercourses but used to be more common.

Introduction

One of the most poignant questions of our time is how to realize a more sustainable use of our remaining natural treasures. There is no easy answer to this question. Ecological, economic and social systems are closely intertwined, resulting in complex, social–ecological systems (SESs). To manage SESs sustainable, we need to ensure that the scientific knowledge effectively meets the information demands of practitioners, who then interactively make scientifically underpinned management decisions. In this thesis, this aspired goal is called "informed science–policy interaction". To reach this goal, we need science–policy interfaces (SPIs), i.e., organizational arrangements for processes of social interaction between scientists and practitioners. Unfortunately, many SPIs encounter interaction problems. Their effectiveness appears to be context-dependent. In addition, the knowledge exchange between scientists and practitioners often needs improvements. Apparently, a clear understanding of how to design and implement them is a persistent gap in our knowledge. This thesis aims to fill this knowledge gap.

The collaborative management process regarding the future of the Dutch peatlands serves as an illustrative context for this endeavor. The overriding problem of this SES is soil subsidence, which results in increasing management costs and the emission of greenhouse gasses. Therefore, several SPIs were instigated that aimed for a more sustainable peatland management. These SPIs have produced generic knowledge, raised awareness of adaptations, and provided guidance for regional adaptation strategies. However, additional SPIs are still needed to support collaborative management of the peatlands at a local level, where stakeholders are faced with the challenge to put generic knowledge into practice.

The main research question is: *"How can SPIs be designed and implemented to advance the collaborative management of SESs?"* To answer this question, an SPI is developed that integrates analytical and interactive decision support tools and embeds them in participatory processes. The resulting SPI is implemented in a real-world case study regarding the future of the Dutch peatlands. The development of the SPI consists of four sequential steps. The first step (Chapter 2) creates an integrated modelling framework which generates spatially and temporally explicit, site-specific knowledge. Steps 2 and 3 (Chapters 3 and 4) combine the modelling framework with a heuristic Cost–Benefit Analysis and a valuation of cultural ecosystem services. Step 4 (Chapter 5) integrates all previous features into an interactive simulation system. The results feed into a general discussion and the overall conclusions of this thesis (Chapter 6). In addition, perspectives are presented regarding the implementation of the SPI and the future of the Dutch peatlands.

Step 1: An integrated modelling framework

The first step focused on an integrated modelling framework that simulated the interrelated dynamics of water management and soil subsidence and that determined the spatial and temporal range of societal impacts. Its core consisted of a GIS-based model that simulated the governing physical processes of the peatlands, i.e., water level dynamics and soil subsidence. Subsequently, stakeholders and scientists co-designed additional GIS models that allowed for an integrated, spatially and temporally explicit assessment of all the relevant impacts of management strategies, which could be distributed to all the stakeholder groups involved.

The framework was used to assess the long-term impacts of water management strategies in the Dutch peatlands. The strategies resulted in marked differences in soil subsidence rates and societal impacts. Because the broad range of assessed impacts, cross-analyses could be made that enriched the understanding of the peatland dynamics. Moreover, the assessments improved awareness of the long-term impacts of management strategies and revealed which stakeholders were disproportionally exposed to these impacts.

The results can make negotiation processes on goals, means, and possible future pathways more transparent. The added value of the modelling framework clearly resulted from its use by the regional water authority of the research area. They cross-analyzed the impacts on real estate damage and water system management, which revealed increasingly unbalanced cost–benefit ratios. The generated insights led the regional water authority to change their current water management strategy, preventing unsustainable future developments.

Step 2: Cost–Benefit Analysis as a heuristic aid

The second step combined the integrated modelling framework with a heuristic Cost–Benefit Analysis, aiming for an intersubjective assessment of management strategies. The combination was used in a collaborative policy process regarding the future of the Dutch peatlands. The stakeholders involved in the case study provided empirical economic data, co-designed the assessments and collaboratively defined water management strategies.

The results exposed that a top-down strategy of raising water levels and achieving a transition in land use was not viable because gains and losses would be unequally distributed. A strategy in which field drains were applied was found to be beneficial for all stakeholders. However, the remaining challenge would be to agree on a fair distribution of the implementation costs. The participating stakeholders discussed all constituent elements of the approach in participatory workshops. The workshops were attended by approximately 240 people, culminating in an exploration of shared interests and context-specific pathways for the collaborative implementation of the management strategies.

The approach was evaluated by 23 real-world stakeholders. They clearly appreciated the comprehensive assessments and the abundant opportunities to discuss the results across multiple perspectives. The approach was deemed well suited to increase the stakeholders' understanding of the complexities of the SES and their ability to explore its potentialities.

Step 3: Participatory non-monetary valuations

The third step revealed how the SPI that was developed in the previous steps could provide a platform to integrate non-monetary valuations in the collaborative management of SESs, enriching the decision-making process. In participatory workshops, several stakeholders jointly designed management scenarios and selected cultural ecosystem service indicators. Subsequently, the non-monetary valuation of the cultural ecosystem service indicators of 295 stakeholders was surveyed.

The results were evaluated in workshops, using a mapping tool that presented spatially explicit valuation maps of cultural ecosystem services from multiple stakeholder perspectives, revealing the spatial consequences of preferences that were not clear beforehand. The participatory mapping approach provided a platform to integrate monetary and non-monetary assessments. In addition, it instigated a discussion among the stakeholders on how differences in transcendental values had shaped their valuation of the research area. If these value deliberations are embedded in a multi-loop social learning process, they may result in the formation of shared cultural values beyond the aggregated utilities of individuals, which may contribute to reaching consensus on collaborative management practices.

Step 4: Interactive simulations

In the fourth step, the SPI created in steps 1–3 was combined with the Tygron Geodesign Platform. This resulted in RE:PEAT, an interactive simulation system for the exploration of collaborative peatland management. As RE:PEAT was applied in ten workshops attended by 89 real-world stakeholders and was evaluated using logfiles, videos, and post-workshop questionnaires, an in-depth perspective was derived that revealed the added value of interactive simulation systems to overcome implementation challenges for collaborative management strategies.

In the workshops, all participants explored not only the physical system dynamics by implementing measures, but also the social system dynamics by brokering deals with other stakeholders. As RE:PEAT translated generic scientific knowledge into site-specific effects from multiple stakeholder perspectives, it created an operational fit between knowledge supply and demand. In addition, because RE:PEAT supported informed negotiations, it also raised awareness of mutually beneficial strategies. As a result, cooperation among the workshop participants was enhanced and their understanding of problems and action perspectives regarding the SES increased.

The research showed that interactive simulation systems should aim for capacity building at the individual and group level, striving for a site-specific awareness of the effects of measures and strengthening the resolve of stakeholders to collectively implement these measures. In addition, the research showed how interventions could be incorporated that prevented individualistic strategies but that fostered cooperative attitudes instead. The embeddedness in preceding SPIs enhanced the credibility and legitimacy of RE:PEAT, whereas its salience was strengthened by abundant detailed information, realistic visual quality, and calculation times that were short enough to keep pace with stakeholder interactions during the decision-making processes.

General discussion and conclusions

The design of the SPI considered seven general principles that were derived from scientific literature. The research improved our understanding of the conditions that enhance a successful implementation of these design principles.

1. The design principle "analyses that are trustworthy, bridge interests and take a multi-perspective focus" was successfully implemented by a participatorily designed, broad scope of the impact assessments, which reflected the stakeholders' interests and values.
2. The design principle "dual accountability to scientists and practitioners" was successfully implemented by the integration of scientific and lay perceptions of the SES, which was permissible because of the heuristic evaluation of multiple evaluative endpoints.
3. The design principle "specialized roles for managing the boundary between science and policy" was successfully implemented by the engagement of experts who improved the understandability of the scientific knowledge from a lay perspective.
4. The design principle "engagement of boundary-spanning participants" was successfully implemented by supporting capacity building and social learning. As a result, the participants of the management process were enabled to play an active role in boundary management, which enhanced its effectiveness.
5. The design principle "use of boundary objects" was successfully implemented by the integration of analytical and interactive decision support tools in an interactive simulation system. The hands-on participation of stakeholders in interactive simulations

created a shared focus with room for science–policy interactions form multiple perspectives, which enhanced both the capacity of the stakeholders to understand the complexities of the social–ecological system and their awareness of options to advance its collaborative management.

6. The design principle "inclusion of a broad coalition of scientists and practitioners" was successfully implemented by the creation of a broad group of scientists and practitioners that ensured that a boundary object became acknowledged in wider social units. This enhanced the ability of the boundary object to support social learning processes and to advance the collaborative management of the SES.
7. The design principle "facilitation of dynamic and interdisciplinary processes" was successfully implemented by heuristic evaluations of the impact assessments. As a result, negotiation processes were enriched by cross-analyses of spatially and temporally explicit assessments from multiple stakeholder perspectives.

The SPI was implemented in the Dutch peatlands. Despite several previous SPIs, the collaborative management in this SES was hampered by operational misfits between the supply and demand of knowledge. The research of this thesis demonstrated how these implementation challenges could be overcome, and the collaborative management of the SES could be enriched by informed science–policy interactions. The SPI is primarily suited to improve operational, site-specific science–policy interactions. Its effectiveness depends on preceding SPIs which have enhanced the generic knowledge of the SES. It is well suited to advance collaborative management processes, but poorly equipped for SESs with centralized governance modes.

Perspectives

The perspectives for further scientific research and for practical applications of the results are aligned. Currently, an active Community of Practice is expanding the features of the Tygron Geodesign Platform, which includes the RE:PEAT template. This will enable highly spatially detailed water quality assessment which can be directly linked to the social–ecological dynamics of the peatlands. It would be interesting to implement and evaluate these additional features. Another interesting development is the longitudinal use of RE:PEAT as a boundary object. Currently, several collaborative management processes in the Dutch peatlands are using the RE:PEAT tool to underpin novel applications of field drains. It would be interesting to explore what is required for RE:PEAT to maintain its usefulness as a boundary object for these processes. Regarding the Dutch peatlands in general, this thesis suggests their resilience can be enhanced by time-and-place-specific trade-offs between restoring biodiversity, decreasing soil subsidence, reducing greenhouse gas emissions, and maintaining agricultural production. Such time-and-place-specific trade-offs will ensure that the peatlands maintain the ability to respond to the challenges that await us in the decades to come. The SPI that was developed in this thesis is specifically tailored to support the interactive negotiation processes with multiple evaluative endpoints that are necessary to deal with complex challenges like this.

Although in this research, the SPI was implemented in the Dutch peatlands, the approach to design and implement an SPI for collaborative management can be repeated in SESs around the world. The bottom line is that decision support tools should be much better tailored to the demands of the collaborative management process, and the participants of this process should be more strongly involved in the design and interactive implementation of the SPI. This will allow us to embrace the complexity of SESs and realize the informed science–policy interactions which are needed to achieve a more sustainable use of our natural treasures.

Samenvatting

For centuries, the rural parts of the Dutch peatlands have primarily been used for dairy farming. This has resulted in a quintessential Dutch landscape, with medieval allotment patterns and the accompanying ditches that still exist (right-hand side of the figure). The meadows with grazing cows (left-hand side of the figure) contribute strongly to the sense of place of the peatlands, whereas the openness of the meadows is valued for its aesthetic value (Chapter 4). The agricultural countryside is also a natural habitat for wildlife such as swallows (right-hand side of the figure) and hares (center of the figure).

Inleiding

Een van de meest dringende vragen van onze tijd is hoe we onze natuurlijke hulpbronnen duurzamer kunnen gebruiken. Helaas bestaat er geen eenvoudig antwoord op deze vraag. Ecologie, economie en maatschappij zijn namelijk nauw met elkaar verweven, wat resulteert in complexe sociaal–ecologische systemen. Om deze complexe systemen duurzaam te beheren is niet alleen wetenschappelijke kennis nodig over hoe het systeem werkt, maar ook over het gedrag van alle gebiedspartijen die het systeem gebruiken. Bovenal moet je zorgen dat de wetenschappelijke kennis ook een nuttige ondersteuning biedt aan de beheerpraktijk van de gebiedspartijen. Daarvoor heb je een Kennis–Beleid Verbinding (KBV) nodig die ervoor zorgt dat wetenschappers en systeemgebruikers samenwerken bij het ontwikkelen en toepassen van kennis. Hoewel er al veel onderzoek gedaan is naar KBV's, treden in de praktijk toch vaak implementatieproblemen op. Daarom beoogt dit proefschrift het inzicht te verbeteren in hoe je KBV's voor een gezamenlijke beleidsontwikkeling voor sociaal–ecologische systemen kan ontwerpen en implementeren. Dit inzicht kan helpen om een goed geïnformeerde interactie teweeg te brengen op het grensvlak tussen kennis en beleid, waardoor een duurzaam beheer van onze natuurlijke hulpbronnen een stap dichterbij komt.

De centrale casus van dit proefschrift betreft het proces van de gebiedspartijen in het Hollandse veenweidegebied om te komen tot een duurzamer beheer van het gebied. Het Hollandse veenweidegebied is een sociaal–ecologisch systeem waarvan het grootste deel van het oorspronkelijke veenpakket al verloren is gegaan, door een combinatie van natuurlijke en antropogene processen zoals zeespiegelstijging, erosie, drooglegging en turfwinning. Het resterende veengebied wordt nu vooral gebruikt voor melkveehouderij. Dat heeft geresulteerd in een sterke zuivelindustrie en het karakteristieke Hollandse cultuurlandschap met zijn bijzondere flora en fauna. Maar het gebied kampt ook met problemen. Door de drooglegging daalt de veenbodem, wat tot steeds hogere beheerkosten leidt en gepaard gaat met de uitstoot van broeikasgassen. Daarom zijn de afgelopen jaren diverse KBV's opgezet om te komen tot een duurzamer beheer van het veenweidegebied. Deze KBV's hebben de kennis vergroot over de veenweiden, de effecten van innovatieve maatregelen en de opties voor regionale adaptatiestrategieën. Echter, om deze inzichten goed te kunnen implementeren is een aanvullende KBV nodig die zich specifiek richt op samenwerking en gebiedsgericht maatwerk.

De centrale vraagstelling van dit proefschrift was: *"Hoe kunnen Kennis–Beleid Verbindingen worden ontworpen en geïmplementeerd die de gezamenlijke beleidsontwikkeling voor sociaal–ecologische systemen verbeteren?"* Om deze vraag te beantwoorden werd een KBV ontwikkeld die bestond uit analytische en interactieve beslissingsondersteunende instrumenten, ingebed in participatieve processen. De KBV werd direct in praktijk gebracht om gebiedsprocessen in het Hollandse veenweidegebied te ondersteunen. De ontwikkeling van de KBV verliep in vier opeenvolgende stappen. In de eerste stap (Hoofdstuk 2) werd een integraal modelraamwerk ontwikkeld dat algemene kennis over het veenweidegebied vertaalde in inzichten met een hoog ruimtelijk en temporeel detailniveau. In stap twee (Hoofdstuk 3) werd het integrale modelraamwerk gecombineerd met een heuristische kosten-batenanalyse. Vervolgens is er in stap drie (Hoofdstuk 4) een waardering van culturele ecosysteemdiensten aan toegevoegd. Het geheel werd in stap vier (Hoofdstuk 5) omgebouwd tot een interactief simulatiesysteem. In alle stappen vond een evaluatie plaats van de meerwaarde die de KBV bood voor de ondersteuning van gezamenlijke beleidsontwikkeling voor sociaal–ecologische systemen. Het proefschrift wordt afgesloten met een overkoepelende discussie en conclusie. Tevens worden aanbevelingen gedaan voor onderzoek en implementatie in het algemeen en voor de toekomst van het Hollandse veenweidegebied in het bijzonder.

Stap 1: Een integraal modelraamwerk

In de eerste stap werd een raamwerk van GIS-modellen ontwikkeld met een hoog ruimtelijk en temporeel detailniveau. De kern bestond uit een GIS-model waarmee de nauw verweven dynamiek van slootpeilen, grondwaterstanden en bodemdaling in samenhang wordt gesimuleerd. Op basis van de uitkomsten zijn vervolgens de langetermijneffecten gesimuleerd voor het watersysteembeheer, het wegbeheer, het rioolbeheer, het onderhoud aan kabels en leidingen, de funderingsschade, de landbouwkundige productie, de natuurkwaliteit, de waardering van het landschap, het aantal recreanten en de uitstoot van CO_2.

Het integrale modelraamwerk werd gebruikt om de langetermijneffecten van een aantal waterpeilstrategieën in beeld te brengen in een deel van het Hollandse veenweidegebied. De onderzochte strategieën bleken te resulteren in uiteenlopende bodemdalingssnelheden en ruimtelijke effectpatronen. Omdat de reikwijdte van de analyses breder was dan in eerdere onderzoeken, konden dwarsverbanden worden gemaakt die het inzicht versterkten in de complexiteit van het veenweidegebied. Tevens werden de langetermijneffecten van waterpeilstrategieën duidelijker. Bovendien bleek welke gebiedspartijen onevenredig werden getroffen door de consequenties.

De inzichten kunnen onderhandelingsprocessen over doelen, middelen en mogelijke handelingsperspectieven transparanter maken. Daardoor kan een nuttige ondersteuning worden geboden aan het verbeteren van beheerstrategieën in veenweidegebieden. Deze meerwaarde werd ook daadwerkelijk in de praktijk aangetoond door het waterschap. Zij gebruikten het modelraamwerk om het nut en de noodzaak van hoogwatervoorzieningen te onderzoeken. Hieruit bleek dat de deze voorzieningen tot langetermijneffecten leidden die zeer onevenredig waren verdeeld over de gebiedspartijen. Dit was voor het waterschap aanleiding om hun beleid voor hoogwatervoorzieningen te wijzigen.

Stap 2: Een heuristische kosten-batenanalyse

De tweede stap combineerde het modelraamwerk met een heuristische kosten-batenanalyse, die was gericht op het uitwisselen van kennis en het vergroten van inzichten. De combinatie werd toegepast in een beleidsverkenning van de toekomst van het veenweidegebied. De gebiedspartijen leverden daarbij empirische data aan, waren medeverantwoordelijk voor de opzet van de kosten-batenanalyse en definieerden gezamenlijk de onderzochte beheerstrategieën.

De onderzochte beheerstrategieën hadden uiteenlopende effecten. De beheerstrategie waarbij het waterpeil niet werd aangepast aan de bodemdaling bleek tot een grootschalige verandering van landgebruik te leiden, waarbij de kosten en baten scheef waren verdeeld tussen de gebiedspartijen. De beheerstrategie waarbij onderwaterdrains werden toegepast bleek daarentegen voor alle gebiedspartijen gunstig, maar wierp wel de vervolgvraag op hoe de kosten verdeeld moesten worden tussen de betrokken partijen. De uitkomsten werden uitvoerig bediscussieerd op bijeenkomsten, die in totaal door circa 240 belangstellenden werden bezocht. De bijeenkomsten leidden tot het besef dat de aanpak van bodemdaling vraagt om samenwerking en gebiedsgericht maatwerk. De gevolgde aanpak werd uitgebreid geëvalueerd door 23 mensen die beroepsmatig bij het veenweidegebied waren betrokken. Zij spraken duidelijk hun waardering uit voor de grondige analyses en de brede discussies over kosten en baten. Geconcludeerd werd dat de gevolgde aanpak de bekwaamheid van de betrokkenen kan vergroten om de complexiteit van een sociaal–ecologisch systeem te doorgronden en de gezamenlijke beheermogelijkheden te verkennen.

Stap 3: Participatieve niet-monetaire waarderingen

In de derde stap werd duidelijk hoe de KBV die was ontwikkeld in de voorgaande stappen de mogelijkheid biedt om de gezamenlijke beleidsontwikkeling voor het veenweidegebied te verrijken met niet-monetaire waarderingen van culturele ecosysteemdiensten. In participatieve workshops werden beheerstrategieën opgesteld en indicatoren gekozen voor culturele ecosysteemdiensten. Via een internet-enquête werd in beeld gebracht hoe 295 belanghebbenden de indicatoren waardeerden. Op basis van de enquêteresultaten werden kaarten gemaakt van de waardering van de beheerstrategieën door agrariërs, bewoners en recreanten. Aanvullend daarop werden ook de kosten en baten van de strategieën berekend. Tenslotte werden alle resultaten tezamen besproken in workshops.

De gevolgde aanpak bleek uitermate geschikt om de belanghebbenden te betrekken bij het opstellen van beheerstrategieën voor het veenweidegebied. Tevens werd duidelijk dat het karakteristieke veenweidelandschap met weilanden, sloten, koeien en vergezichten in het algemeen hoog werd gewaardeerd, maar dat agrariërs, bewoners en recreanten niet alle aspecten hetzelfde waardeerden. Bovendien werd de ruimtelijke situering van de waardering inzichtelijk. In de afsluitende workshops bleek de inzichtelijk ontsloten combinatie van monetaire en niet-monetaire informatie heel geschikt om ondersteuning te bieden aan de oriëntatie op een gezamenlijke strategie om een duurzamer beheer van het veenweidegebied in praktijk te brengen. Bovendien werd een aanzet gemaakt tot de vorming van gemeenschappelijke waarden, wat uiteindelijk kan bijdragen aan het overbruggen van tegenstellingen en het in praktijk brengen van gemeenschappelijke beheerstrategieën.

Stap 4: Interactieve simulatie

In de vierde stap werd de KBV uit de voorgaande stappen gecombineerd met het Tygron Geodesign Platform. Het resultaat was RE:PEAT, een interactief simulatiesysteem voor de evaluatie van gezamenlijke beheerstrategieën in veenweidegebieden. RE:PEAT werd toegepast in tien workshops, die werden bijgewoond door 89 mensen die beroepsmatig bij het veenweidegebied waren betrokken. Om een gedegen inzicht te verkrijgen in de meerwaarde van RE:PEAT voor de ondersteuning van het veenweidebeheer, werden vragenlijsten, logfiles en filmbeelden van de workshops in samenhang geanalyseerd.

De deelnemers aan de workshops onderzochten zowel de dynamiek van het fysieke veenweidesysteem, door het uitproberen van maatregelen, als de mogelijkheden van het sociale systeem, door te onderhandelen met andere deelnemers. RE:PEAT vertaalde de algemene veenweidekennis in gebiedspecifieke inzichten vanuit het perspectief van alle belanghebbenden, waardoor het goed aansloot bij de informatiebehoefte van de deelnemers. Tevens werden de deelnemers zich meer bewust van wederzijds voordelige strategieën, waardoor ze zeer duidelijk aangaven dat zowel hun samenwerkingsbereidheid als hun kennis over het veenweidegebied werd versterkt.

Het onderzoek toonde aan dat interactieve simulatiesystemen vooral inzicht moeten bieden in de gebiedspecifieke effecten van maatregelen en daarnaast stimulerend moeten zijn voor de bereidheid van de deelnemers om deze maatregelen gezamenlijk te implementeren. Tevens bleek dat de samenwerkingsbereidheid van de deelnemers versterkt kan worden door interventies in te bouwen die leiden tot discussie over maatregelen. De kwaliteit van een interactief simulatiesysteem bleek afhankelijk van de inbedding in voorafgaande KBV's, een ruim aanbod aan gedetailleerde informatie, goede visuele kwaliteit en berekeningen die snel genoeg zijn om het interactieve besluitvormingsproces bij te houden.

Discussie en conclusie

Bij de ontwikkeling van de KBV voor de gezamenlijke beleidsontwikkeling voor sociaal–ecologische systemen werden zeven algemene ontwerpbeginselen gebruikt uit de wetenschappelijke literatuur. Het onderzoek verbeterde het inzicht in de condities die bijdragen aan een succesvolle implementatie van deze ontwerpbeginselen.

1. Het ontwerpbeginsel 'analyses die betrouwbaar zijn, tegenstellingen overbruggen en meerdere invalshoeken belichten' werd succesvol geïmplementeerd door de kennis van de belanghebbenden te gebruiken om analyses te ontwerpen met een brede reikwijdte en een hoog detailniveau in ruimte en tijd. Hierdoor omvatten de analyses alle aspecten die de belanghebbenden relevant vinden.
2. Het ontwerpbeginsel 'dubbel verantwoording afleggen over kennis, richting zowel wetenschappers als belanghebbenden' werd succesvol geïmplementeerd door een heuristische opzet van de KBV, die meerdere evaluatiecriteria beschouwde zonder hierin een rangorde voor te schrijven. Daardoor konden de interesses van de belanghebbenden volledig worden meegewogen, zonder afbreuk te doen aan de kwaliteit van de gebruikte methoden.
3. Het ontwerpbeginsel 'investeren in communicatie, vertaling en bemiddeling tussen kennis en beleid' werd succesvol geïmplementeerd door het betrekken van specialisten die de wetenschappelijke kennis beter begrijpelijk maakten voor belanghebbenden.
4. Het ontwerpbeginsel 'inschakelen van bemiddelaars tussen de wereld van kennis en beleid' werd succesvol geïmplementeerd door de bekwaamheid van de belanghebbenden om zelf te bemiddelen tussen kennis en beleid te vergroten.
5. Het ontwerpbeginsel 'gebruik van 'grensobjecten' die een gezamenlijke focus bieden, maar ook ruimte laten voor uiteenlopende interpretaties' werd succesvol geïmplementeerd door analytische en interactieve beslissingsondersteunende instrumenten te combineren in een interactief simulatiesysteem. Daarmee konden alle belanghebbenden kennis–beleid interacties aangaan vanuit hun eigen invalshoek, met behoud van een gezamenlijke focus. Hierdoor gingen ze de complexiteit van het sociaal–ecologische systeem beter begrijpen en werden ze zich bewust van gemeenschappelijke strategieën voor een duurzamer beheer.
6. Het ontwerpbeginsel 'betrekken van een brede coalitie van wetenschappers en belanghebbenden die gezamenlijk kennis ontwikkelen, terwijl ze in verbinding blijven met de beleidspraktijk' werd succesvol geïmplementeerd door het opzetten van een brede 'Community of Practice' die zich richtte op het verbeteren van een 'grensobject', dat daardoor steeds beter ondersteuning kon bieden aan het opstellen van gezamenlijke beheerstrategieën en in steeds bredere kring werd toegepast.
7. Het ontwerpbeginsel 'faciliteren van een dynamische, interdisciplinair proces van productie en uitwisseling van kennis' werd succesvol geïmplementeerd door het heuristisch evalueren van alle bestanddelen van de resultaten, waardoor het onderhandelingsproces verrijkt werd met dwarsverbanden vanuit meerdere invalshoeken en de gevolgen van beheerstrategieën inzichtelijk werden.

De KBV werd toegepast in het Hollandse veenweidegebied. Het onderzoek toonde hoe de gezamenlijke beleidsontwikkeling voor dit sociaal–ecologische systeem verrijkt kon worden met geïnformeerde kennis–beleid interacties. Een succesvolle toepassing in andere sociaal–ecologische systemen is mede afhankelijk van de mate waarin voorafgaande KBV's de algemene kennis over die systemen hebben vergroot. Daarnaast is de KBV vooral gericht op gezamenlijke beleidsprocessen, maar minder geschikt voor centraal aangestuurde, niet-participatieve strategieën.

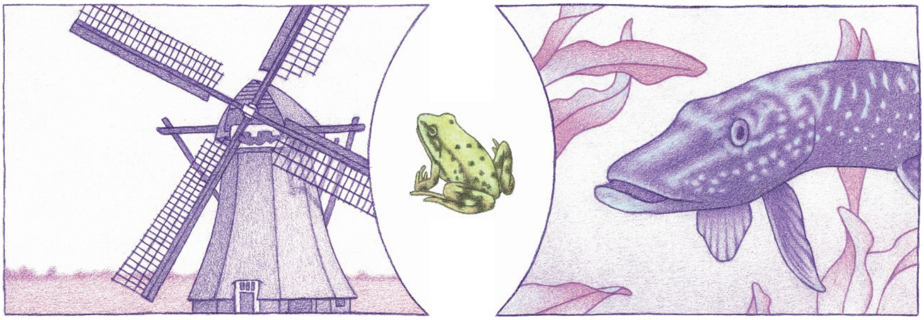
Aanbevelingen

Het onderzoek in dit proefschrift richtte zich op zowel het ontwerp als de implementatie van een KBV. Deze lijn is voortgezet in de aanbevelingen, die vooral zijn gericht op het verder versterken van de bruikbaarheid van de KBV. Deze aanbevelingen maken derhalve geen onderscheid tussen enerzijds de wetenschap en anderzijds de beleidspraktijk.

- Momenteel ontwikkelt een actieve Community of Practice aanvullende analysemogelijkheden voor het Tygron Geodesign Platform, waar RE:PEAT een onderdeel van is. Deze aanvullingen beogen een gedetailleerde ruimtelijke analyse van waterkwaliteit en ecologie te verbinden met de menselijke handelingen die het sociaal–ecologische systeem beïnvloeden. Het zou interessant zijn om de meerwaarde hiervan te onderzoeken.
- Momenteel wordt in diverse gebiedsprocessen praktijkgericht onderzocht welke mogelijkheden gezamenlijk adaptaties zoals onderwaterdrains en paludicultuur bieden voor een duurzamer veenweidebeheer. Het zou interessant zijn om te onderzoeken wat ervoor nodig is om de bruikbaarheid van de in dit proefschrift ontwikkelde KBV gedurende de gehele looptijd van deze gebiedsprocessen te waarborgen en verder te versterken.
- Interactieve simulatiesystemen zullen in toenemende mate aan belanghebbenden de mogelijkheid bieden om zelf analyses uit te voeren. Hierdoor zullen de experts die traditioneel deze analyses verrichtten zich meer moeten gaan toeleggen op het ontwikkelen en ontsluiten van nieuwe kennis waarmee kennisvragen beantwoord kunnen worden die door de interactief simulerende belanghebbenden worden opgeworpen. Het zou interessant zijn om te onderzoeken hoe invulling gegeven kan worden aan deze nieuwe rollen.
- Met betrekking tot het Hollandse veenweidegebied is de aanbeveling dat per gebied alle belanghebbenden gezamenlijk een integrale afweging moeten maken tussen het herstellen van de biodiversiteit, het verminderen van bodemdaling, het minimaliseren van broeikasgasemissies en het faciliteren van een natuurinclusieve landbouw. Dit gebiedsgerichte maatwerk moet waarborgen dat het veenweidegebied als geheel de veerkracht behoudt om in de komende decennia in te spelen op de voorziene en onvoorziene uitdagingen waarvoor we gesteld komen te staan. Gelukkig is de in dit proefschrift ontwikkelde KBV specifiek toegespitst op het ondersteunen van dergelijke complexe uitdagingen.

Hoewel dit onderzoek zich specifiek richtte op het Hollandse veenweidegebied, kan de manier van ontwerpen en implementeren van een KBV voor gezamenlijke beleidsontwikkeling worden herhaald in sociaal–ecologische systemen over de gehele wereld. De essentie is dat beslissingsondersteunende instrumenten veel beter toegesneden worden op de informatiebehoefte vanuit het gezamenlijke beleidsproces en dat de deelnemers aan dit proces veel nauwer betrokken worden bij het ontwerpen en interactief implementeren van de KBV. Hierdoor kunnen we de complexiteit van sociaal–ecologische systemen omarmen en de geïnformeerde kennis–beleid interacties realiseren die nodig zijn om onze natuurlijke hulpbronnen duurzamer te beheren.

Nawoord



In the Middle Ages, the natural fens of the Dutch peatlands were drained, to make them suited for agriculture. This required an intricate water system, encompassing several hundreds of thousands of kilometers of watercourses and tens of thousands of pumps, weirs, and culverts. In the figure, this is depicted by the traditional windmill (left-hand side). The surface water levels have to be lowered periodically, to compensate for the continuing soil subsidence. In addition, the management of the water quality is also needed. In the figure, this is depicted by the pike (right-hand side) and the frog (center).

Jong geleerd, oud gedaan

Ik ben opgegroeid op de Waterman in Veenendaal, wat wellicht een voorteken was dat ik later bij waterschappen de veenbodemdaling zou gaan onderzoeken. Hoe dat ook moge zijn, ik moet in ieder geval constateren dat de aanloop naar mijn proefschrift al is begonnen in de jaren '80. Mijn moeder was toentertijd namelijk natuurgids, waardoor ik het volkomen vanzelfsprekend vond om na schooltijd op strooptocht te gaan langs alle slootjes in de buurt, gewapend met schepnetten en determinatiekaarten. Mijn fascinatie voor hydrologie en aquatische ecologie moet toen al ontstaan zijn. Mijn vader was eveneens een natuurliefhebber. In het weekend nam hij ons graag mee op een fietstocht door het Binnenveld, het veenweidegebied tussen Veenendaal en Wageningen. Ik herinner me hoe ik als klein kereltje een keer vol verbazing aan hem vroeg waarom het daar altijd zo'n kabaal was. Geamuseerd legde hij uit dat de weidevogels dat deden, de Grutto's en Tureluurs die verderop voorbij vlogen en vooral dat kleine beestje dat daar hoog in de lucht luidkeels rondfladderde, de Veldleeuwerik. Waarschijnlijk is toen al mijn fascinatie voor het veenweidegebied begonnen. **Pa en ma**, hartelijk dank voor deze mooie basis! Het was heel fijn dat jullie nog hebben meegemaakt dat ik in 2012 met mijn promotieonderzoek begon, maar ook ontzettend jammer dat jullie allebei de afronding daarvan in 2019 niet meer mee konden maken.

Tussen wetenschap en waterschap

Vanuit deze basis ben ik Milieukunde en Fysische Geografie gaan studeren aan de Universiteit Utrecht en vervolgens gaan werken als hydroloog bij diverse waterschappen. Toen ik in 2010 aanbelandde bij het Hoogheemraadschap De Stichtse Rijnlanden, kreeg ik de ruimte om eens goed mijn tanden in het veenweidedossier te zetten. We vatten het ambitieuze plan op om een afwegingsystematiek te ontwikkelen voor de kosten en baten van peilbesluiten en watergebiedsplannen, met als basis een serie GIS-modellen die de langetermijneffecten in kaart moesten brengen van peilbeheer en bodemdaling. En dankzij diverse GIS-vaardige collega's bleek dat nog te lukken ook. Dankjewel **Harm, Epke, Astrid, Martin en Jan Willem**, voor het enthousiasme waarmee jullie deze uitdaging met me zijn aangegaan. Achteraf bleek dit het officieuze startpunt van mijn promotieonderzoek. Zonder jullie was het me nooit gelukt om al die jaren zulk mooi onderzoek te doen!

Dat er überhaupt sprake kon zijn van een promotieonderzoek kwam doordat mijn collega Peter en ik een keer vrijblijvend de samenwerkingsmogelijkheden met de Universiteit Utrecht waren gaan bespreken. Dankjewel **Peter**, voor de vanzelfsprekendheid waarmee je samen met mij dit gesprek bent aangegaan. Die bijeenkomst bleek te resulteren in een jarenlange samenwerking. Jerry en Paul besloten namelijk vanuit de universiteit met ons mee te denken over de opzet van onze afwegingsystematiek. Wat aanleiding vormde voor uitgebreide veenweidebespiegelingen, waar Jerry en ik eigenlijk nooit mee opgehouden zijn. Dankjewel **Jerry**, voor die nooit ophoudende interesse. Paul kreeg de smaak ook goed te pakken en stelde voor om de samenwerking voort te zetten via een promotieonderzoek. Dankjewel **Paul**, voor dat initiatief en voor alle minutieuze opbouwde feedback die je me sindsdien als copromotor hebt gegeven. En **Kees**, ook jij bedankt voor de ruimte en de ondersteuning die je gaf om vanuit het waterschap deze onderneming mogelijk te maken.

In de zomer van 2012 schreef ik een Plan van Aanpak vol wilde ideeën voor kennisproducten die de participatieve planvorming van het waterschap effectiever moesten gaan ondersteunen. Daarmee wist Paul de interesse te wekken van Martin en Peter om zich als promotor aan het beoogde onderzoek te verbinden. Tijdens een eerste oriënterende bespreking merkte Martin lichtjes geamuseerd op dat er een persoonlijke frustratie doorklonk in het plan, over de in mijn ogen haperende kennisbenutting in de waterschappraktijk. Dat leek hem een uitstekend vertrekpunt voor een promotieonderzoek. Peter leek het ook interessant, met de kanttekening dat de inbedding in de wetenschappelijke literatuur nog wel enigszins verhelderd diende te worden. Een terecht punt, zo bleek, want zeven jaar later worstelde ik daar nog steeds mee. Maar de kogel was door de kerk. Per 1 november 2012 mocht ik beginnen met het promotieonderzoek. **Martin en Peter**, dankjewel dat jullie me die kans boden. En bovendien dat jullie nooit zijn opgehouden met aan me te vragen waar ik nou eigenlijk naartoe wilde, wat ik precies bedoelde en hoe ik de inbedding in de wetenschappelijke literatuur zag.

<u>De veenweiden in</u>

Mijn onderzoek bevond zich op het snijvlak van wetenschap en praktijk en tevens op het snijvlak van natuurwetenschappelijke en maatschappijwetenschappelijke kennis. Wat het erg boeiend maakte, maar waardoor het ook alle kanten op kon gaan. Bijvoorbeeld, het evalueren van planvormingsprocessen, wat weliswaar erg interessant was, maar wat ik uiteindelijk toch niet heb gebruikt in mijn proefschrift. Desalniettemin, dankjewel **Corina** voor het enthousiasme waarmee je deze processen samen met me evalueerde. Een ander voorbeeld betrof waterkwaliteit. Paul en ik hadden een ambitieus plan opgevat voor een snel rekenend, visueel geavanceerd beslissingsondersteunend systeem waarmee je in elk slootje op elk moment de waterkwaliteit inzichtelijk kon maken. Een paar jaar lang stuurden we ook vol enthousiasme studenten het veld in om alvast metingen te verrichten waarmee we dit systeem zouden kunnen ijken. Helaas bleek het plan uiteindelijk te ambitieus om echt tot uitvoer te brengen. Desalniettemin, dankjewel **Nadia, Susy, Nikki, Gea, Emmy, Madeleine, Jan Willem, Florian, Marieke en Vanya**, voor alle metingen en analyses, **Rob**, voor het begeleiden van het veldwerk en **Bas en Yolanda,** voor het meedenken en -dromen over de waterkwaliteitsambities.

Uiteindelijk kozen we ervoor de focus helemaal te leggen op het veenweidegebied. We pakten de draad op van de reeds ontwikkelde GIS-modellen, brachten enkele verbeteringen aan en implementeerden ze in beleidsprocessen over hoogwatervoorzieningen en bodemdaling. Dankjewel **Linda**, voor het samen optrekken bij de boeiende hoogwatervoorzieningenworsteling, **Martin, Jan, Ad, Harm** en alle andere betrokkenen bij de 'Toekomstverkenning bodemdaling', voor de fijne samenwerking en het aanzwengelen van een goed geïnformeerde discussie over bodemdaling, **Ernst en Theo**, voor de hulp bij de kosten-batenanalyses, **Daan en Len**, voor het verder verbeteren van de analyses, **Co**, voor alle prachtige illustraties, en natuurlijk **Harm, Epke, Astrid en Jan Willem**, voor alle berekeningen met de GIS-modellen. Daarnaast wil ik graag **Patrick en Jan** bedanken voor de samenwerking bij de beleidsdiscussies op het bestuurlijke niveau. Het was heel boeiend om jullie te kunnen ondersteunen bij jullie inzet voor gebiedsgericht maatwerk en een constructieve samenwerking tussen alle gebiedspartijen.

Toen ik naar aanleiding van de 'Toekomstverkenning bodemdaling' een lezing mocht houden bij de Provinciale Adviescommissie Leefomgevingskwaliteit in Zuid-Holland ontstond het idee om een methode te ontwikkelen om cultuurhistorie mee te kunnen wegen in kosten-batenanalyses. Die uitdaging kon ik natuurlijk niet laten liggen, dus enkele maanden later begonnen we met het 'Waarderingsonderzoek veenweide'. Dankjewel **Mathijs, Michel, Merten, Welmoed en Harm**, voor het enthousiasme waarmee jullie dit idee gelijk omarmden. Het was een prachtig project, waarbij ik in korte tijd veel wijzer ben geworden over cultuurhistorie en landschapswaardering, vooral dankzij de 'colleges' van Michel tijdens de bijeenkomsten van onze projectgroep.

Het ‘Waarderingsonderzoek veenweide’ zette de lijn voort van het inbedden van gedegen analyses in uitvoerige discussies. De eerste discussiebijeenkomst was gelijk op de mooiste cultuurhistorische plek die het Hoogheemraadschap De Stichtse Rijnlanden te bieden had: de oude bestuurskamer in het Dijkhuis te Jaarsveld. Dank aan alle deskundigen die deelnamen aan die discussie en gezamenlijk ontrafelden wat de totaalbeleving van de veenweiden was: meer dan ‘het casco’ van sloten en weilanden, maar ook ‘de stoffering’ van de cultuurhistorie en de ecologie. De discussie werd vervolgd op het internet, dankzij de mooie website, kaarten en illustraties die **Frank** en zijn collega’s hadden gemaakt. Daarna op de Koeienmarkt in Woerden, dankzij het enthousiasme van **Monica, Welmoed, Arend, Bart, Astrid, Astrid, Harm en Linda**. En ten slotte in de workshops met DialogueMaps. Dank aan iedereen die daar een succes van maakte. Het heeft me laten inzien hoe belangrijk het is om niet alleen over cijfers en getallen te spreken, maar ook over de waarden die onze beleving en overtuiging kleuren.

<u>Hoogtepunten</u>

Hoe interessant alles tot dan toe ook was geweest, het hoogtepunt moest nog komen. Ik liep toentertijd rond met een vaag omlijnd plan voor een interactief dashboard ter ondersteuning van de discussies over de onderzoeksresultaten. Simon kwam toen met het ludieke idee om met Tygron in gesprek te gaan, een bedrijf dat ‘serious games’ maakten. Ik had geen idee wat dat precies betekende, maar het kostte minder dan vijf minuten toelichting en ik was helemaal om: een serious game over bodemdaling zou het interactieve sluitstuk van mijn onderzoek worden! Dankjewel **Simon**, voor dit idee. En natuurlijk dank aan **Florian, Raymond, Rudolf, Joris, Maxim, Bobby, Frank, Hansje, Hedi** en alle andere Tygron-medewerkers, die het idee hebben omgevormd tot een klinkend succes. Jullie aanpak is verfrissend vooruitstrevend en wat mij betreft precies die opkikker die het Nederlandse waterbeheer nodig heeft.

De Tygron-techniek was misschien fantastisch, door de mensen werd het prachtig. Dankjewel **Simon, Ernst, Jan, Gert Jan, Gilles, John, Welmoed, Gert, Kees en Max**, voor het meedenken over het ontwerp van RE:PEAT. Dank ook aan alle deelnemers van de RE:PEAT sessies. Elke sessie was voor mij een hoogtepunt, maar de vuurdoop van RE:PEAT, op 10 maart 2016 in Het Oude Station van Houten, spande de kroon. Vanaf dat moment wist ik namelijk dat interactieve simulatie ook echt werkte en dat mijn promotieonderzoek geslaagd was. Meer dan speciale dank gaat uit naar de collega’s die keer op keer een bijdrage leverden aan alle RE:PEAT sessies. Dankjewel **Harm, Epke, Astrid en Marije**, voor jullie hulp, waardoor de RE:PEAT sessies niet alleen een succes werden, maar ook een feest om mee te maken. Dank ook aan **Bert**, die als hoogheemraad vaker dan wie dan ook heeft deelgenomen aan RE:PEAT sessies en telkens weer heeft laten zien welke mooie resultaten je kan halen als je de kunst verstaat van het polderen. Als herinnering aan deze periode heb ik de achterkant van dit proefschrift versiert met een tekening van onze eerste sessie.

<u>Wie schrijft, die blijft (schrijven)</u>

Daarmee was het onderzoek klaar, maar het proefschrift nog niet. Ik moest het namelijk ook nog allemaal opschrijven en gepubliceerd zien te krijgen. Dat bleek het moeilijkste, maar ook het meest leerzame aspect van het hele traject. Het begon nog voorspoedig, met bijdragen aan gepubliceerde papers van Nikki et al. (2013) over fosfaatnalevering en Jerry et al. (2015) over weidevogels. Paul sprak in die periode ook nog optimistisch over resultaatvoetbal, dat we ‘even’ een eerste paper gingen publiceren over de ‘Toekomstverkenning bodemdaling’, waar we eind 2013 al bijna mee klaar waren. Helaas bleken mijn schrijfpogingen net zo onsuccesvol als de resultaten van het Nederlands voetbalelftal in die periode. Uiteindelijk heb ik twintig verschillende versies van de beoogde paper geschreven en zeven pogingen ondernomen om hem gepubliceerd te krijgen, bij vier afzonderlijke tijdschriften. Maar eind goed, al goed: bijna vijf jaar later, in juni 2018, lukte het uiteindelijk toch. Dankjewel **Peter, Paul en Martin**, dat jullie nooit zijn afgehaakt en al die tijd steeds weer opbouwende feedback zijn blijven geven.

De omslag in deze Processie van Echternach kwam door de suggestie van Paul om er twee papers van te maken, eentje over de GIS-modellen en eentje over de kosten-batenanalyse. Daardoor ging het een stuk sneller: de modellen-paper kostte 'slechts' negen versies, met een doorlooptijd van iets meer dan twee jaar. In die tijd leerde ik langzaamaan hoe je als wetenschapper je onderzoek kan ontleden en op een nog pakkendere manier voor het voetlicht kan brengen. Een kwestie van stug blijven volhouden, zelfs als dat volgens de gangbare opinie toch wel een beetje erg zot begint te worden. Of, zoals Martin het verwoordde: "Een wetenschapper is slim, creatief, werkt hard, maar moet vooral ook volhouden vanuit de overtuiging dat ie iets belangrijks te melden heeft." Eigenlijk zijn het dus net beeldhouwers …

Daarna ging het sneller, begon ik te snappen hoe je kan 'beeldhouwen' met wetenschappelijke teksten. De vierde paper lukte zelfs in slechts zes versies, met maar een jaar doorlooptijd vanaf de eerste versie tot de publicatie. Al met al het resultaat van zeven jaar academische vorming: het lezen van honderden papers, het presenteren op symposia zoals Alter-Net (Gent, 2017), Resilience (Stockholm, 2017), LANDac (Utrecht, 2017), Pathways to Sustainability (Utrecht, 2018) en Water Science for Impact (Wageningen, 2018), maar vooral het spreken met en luisteren naar alle bevlogen wetenschappers in het Copernicus Instituut. Hartelijk dank aan iedereen voor die zeven mooie jaren. Met speciale dank aan **Armin, Esther, Hans, Karin en Martin**, voor het verkennen van een mogelijke bijdrage aan de Hub 'Water, Climate & Future Delta's', en tevens aan **Eric**, voor de bereidheid om daar vanuit het waterschap aan mee te werken. Daarnaast wil ik ook graag **Joy en Fulco** bedanken, voor alle heldere aanwijzingen om te komen tot teksten in 'unclogged English'.

<u>Het belangrijkste</u>

1 november 2019, als ik op de dag af zeven jaar met mijn promotie bezig ben, komt er een einde aan die ontdekkingstocht. Dankjewel **Carolien, Hens, Jos, Wil en Wolf**, dat jullie mijn thesis kritisch hebben doorgelezen en me de kans geven om hem te verdedigen. Als ik tegenover jullie sta, word ik geflankeerd door twee paranimfen waar ik ontzettend trots op ben. Ze staan namelijk allebei voor een grote groep mensen die mij bijzonder hebben geholpen en geïnspireerd. Dankjewel **Harm en Florian**, dat jullie mij willen vergezellen tijdens deze belangrijke bijeenkomst. Dat betekent veel voor me!

Harm, jou heb ik gevraagd omdat je zeven volle jaren, met een niet aflatend enthousiasme hebt meegewerkt aan al mijn onderzoeken. Een betere collega kon ik echt niet wensen! Ik heb je mede gevraagd om **Astrid en Epke** te vertegenwoordigen, die bijna net zo enthousiast betrokken zijn geweest bij alle onderdelen. En daarnaast ook alle andere collega's bij het Hoogheemraadschap De Stichtse Rijnlanden en daarbuiten die hebben meegewerkt aan het uitvoeren van het onderzoek en de verdere doorwerking ervan in de waterschappraktijk. Speciale dank gaat uit naar **Ko**, voor het aanstekelijke enthousiasme waarmee je iedereen bewust maakte van broeikasgasemissies, naar **Erik, Gert Jan, Gilles en Roelof** voor de deskundigheid waarmee jullie mijn onderzoek, de waterschappraktijk en het veenweidedebat in het algemeen bleven voeden, naar alle betrokkenen bij het onderwaterdrainageproject in polder Lange Weide, en naar **Chris en Gert** in het bijzonder, voor de ongebreidelde energie waarmee jullie als tweepersoons Kennis–Beleid Verbinding dat project tot een van de meest succesvolle veenweideprojecten van de afgelopen jaren wisten om te toveren.

Florian, jou heb ik gevraagd omdat ik dankzij de Tygron software mijn onderzoek naar een hoger niveau kon tillen, mijn ideeën nog verder kon uitwerken en daadwerkelijk kon implementeren in de waterschappraktijk. Ik heb je mede gevraagd om al je Tygron-collega's en alle betrokkenen bij onze Community of Practice te vertegenwoordigen. Het was waanzinnig mooi om mee te maken hoe binnen een paar jaar tijd zoveel organisaties van jullie watersoftware gebruik zijn gaan maken. Hartelijk dank aan iedereen die daar een rol in heeft gespeeld, in het bijzonder de enthousiastelingen bij de STOWA, de provincie Zuid-Holland en het programmabureau Utrecht-West die de ontwikkeling van RE:PEAT en het vervolg daarop mogelijk maakten. Tevens alle lof voor **Jantine, Joost en Epke**, voor de gezamenlijke organisatie van een van de beste symposia waar ik ooit geweest ben: 'Inundatie? Reken maar!', voor alle Waternet-collega's die ook met de Tygron-software aan de slag willen, en voor **Harm, Jan, Joost, Thijs, Jan Jelle, Bruce, Martine en Marijke**, voor hun bijdrage aan de ontwikkeling van de watermodule. En natuurlijk een dikke grijns richting **Sven, Bob en Wolf**, voor de eerste stappen op weg naar een integratie met PCDitch. Stel je toch voor, binnenkort krijgen we dus alsnog een snel rekenend, visueel geavanceerd beslissingsondersteunend systeem waarmee je in elk slootje op elk moment de waterkwaliteit inzichtelijk kan maken. Kicken!

Mijn nawoord is opzettelijk een lang relaas geworden, om aan te geven dat promoveren niet alleen draait om onderzoek en publicaties, maar ook valt en staat met alle mensen om je heen. En dan heb ik de belangrijksten nog niet eens genoemd: vrienden en familie. Hartelijk dank aan jullie allemaal, omdat jullie me de afgelopen jaren keer op keer weer succes bleven wensen. Met meer dan speciale dank aan **Betsie, Wim en Tonnie**, die dat vaker dan wie ook hebben gedaan, aan **Julia**, die jarenlang belangstellend bleef vragen of mijn veenweidespel al af was (wat ik zelf ook bijna net zo belangrijk vond als de afronding van het proefschrift zelf), aan **Manja en Pepijn**, die zich volledig konden inleven in de alles ontwrichtende worsteling die het af en toe was, aan jaarclub **Multiplicamini**, die al bijna drie decennia getuige is van al mijn studieperikelen, en aan **Bernie, André en Hielke**, die zelf ook naast hun werk een studie deden, maar toch nog regelmatig tijd wisten te vinden voor gezamenlijke bierproeverijen.

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Curriculum Vitae

The Dutch peatlands are a major European stronghold for meadow birds. In the figure, this is depicted by the Eurasian oystercatcher and the flight of northern lapwings and Eurasian curlews (left-hand side). The area is well-suited for breeding and chick rearing because of its openness, which enhances the visibility of predators such as the western marsh harrier (center). The population of meadow birds in the Dutch peatlands has markedly declined during the previous decades. Simultaneously, the population of geese has increased strongly. For example, the Canada goose (right-hand side).

About the author

Henk van Hardeveld (1973) studied Environmental Sciences and Physical Geography at Utrecht University. His master theses focused on the implementation of co-management of natural resources (1997) and the hydrological analysis of a tidal wetland (2001). Building on these experiences, he worked at the interface of hydrological research and participatory policy processes at several Dutch water authorities (Hoogheemraadschap van Rijnland, 2001–2006; Hoogheemraadschap van Delfland, 2006–2010; Hoogheemraadschap De Stichtse Rijnlanden, 2010–2019). In 2012, he began a PhD at the Copernicus Institute of Sustainable Development at Utrecht University, under the supervision of Martin Wassen, Peter Driessen and Paul Schot. Using the water management of the Dutch peatlands as an illustrative context, his research aimed to further advance the support of collaborative management of social–ecological systems. The results of his research are presented in this thesis. Currently, he is the manager of a team of hydrologists and ecologists at the Research and Consultancy Department at Waternet. In his spare time, he sculpts, draws, and develops boardgames. Extracts of his latest boardgame, which is about the complexities of peatland management, are presented throughout this thesis. He lives in Huizen with his girlfriend Wendy and their cats Daantje and Dobby.

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To manage our social–ecological systems more sustainable, collaborate management processes are often advocated, supported by Science–Policy Interfaces that reconcile the supply of scientific information with users' demands. Unfortunately, many Science–Policy Interfaces encounter interaction problems that diminish their effectiveness. Therefore, this thesis presents a novel approach to design and implement Science–Policy Interfaces, which can guide us towards the informed science–policy interactions that are needed to manage our social–ecological systems more sustainable.

The general research approach is to integrate analytical and interactive decision support tools, embed them in participatory processes, and implement and evaluate the resulting Science–Policy Interface in a real-world case study regarding the future of the Dutch peatlands. The research demonstrates how to enhance cooperation among stakeholders, increase their understanding of the complexities of social–ecological systems, and raise their awareness of collaborative options to manage these systems more sustainable.