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

H.A. van Hardeveld[a,b,*], P.P.J. Driessen[a], P.P. Schot[a], M.J. Wassen[a]

[a] Copernicus Institute of Sustainable Development, Utrecht University, P.O. Box 80.115, 3508 TC Utrecht, The Netherlands
[b] Regional Water Authority Hoogheemraadschap De Stichtse Rijnlanden, P.O. Box 550, 3990 GJ Houten, The Netherlands

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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 Dutch peatlands. Average soil subsidence rates were shown to range from 0.6 to 4.5 $mm \cdot y^{-1}$, resulting in marked differences in societal impacts that affect stakeholders unequally. Moreover, the impacts on real estate damage and water system maintenance 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.

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).

* Corresponding author at: Copernicus Institute of Sustainable Development, Utrecht University, P.O. Box 80.115, 3508 TC Utrecht, The Netherlands.
E-mail address: h.a.vanhardeveld@uu.nl (H.A. van Hardeveld).

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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. Methods

2.1. Research area

We focus on a part of the peatlands in the western part of The Netherlands, covering 440 km^2 (Fig. 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.

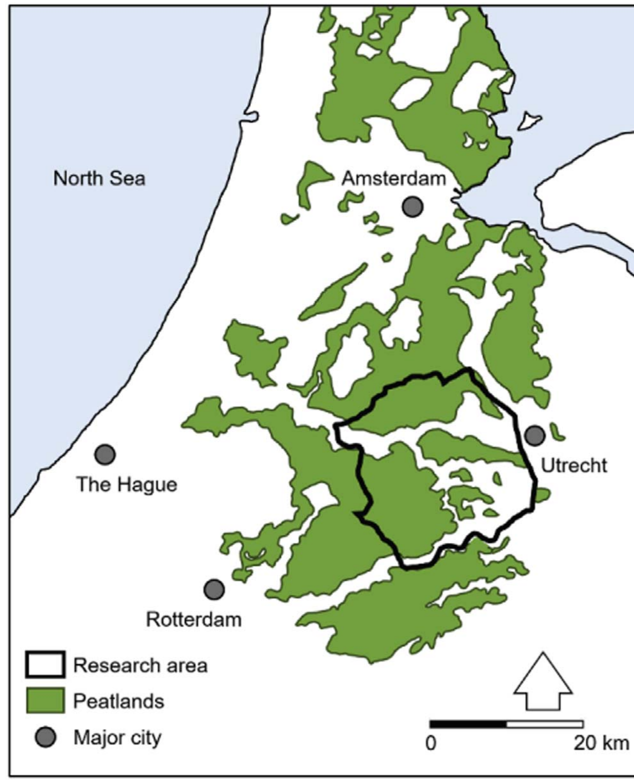

Fig. 1. Location of the research area in the western part of The Netherlands.

2.2. Management strategies explored on their impacts

We used our integrated modelling framework (Fig. 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

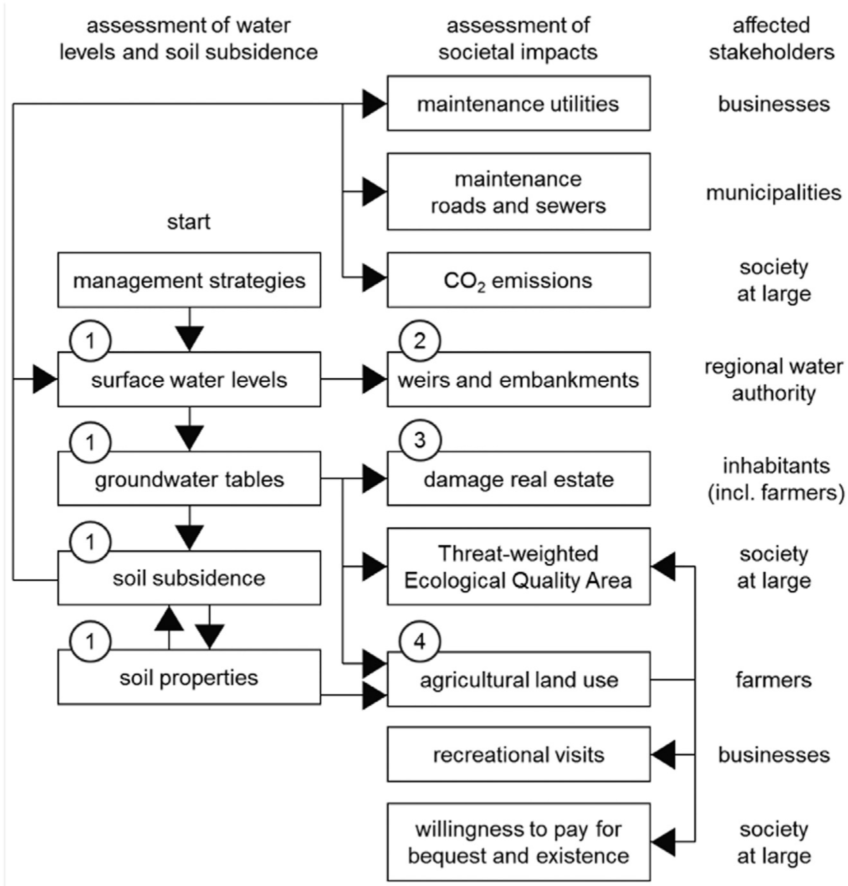

Fig. 2. Design of the integrated modelling framework used in the current study. Arrows indicate the sequence of the assessments. Numbered circles indicate the GIS-models we developed.

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.

2.3. Design of water levels and soil subsidence assessment

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) for these reasons too. See Appendix 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 strategies 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 strategies 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, 1986). 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 Eq. (1).

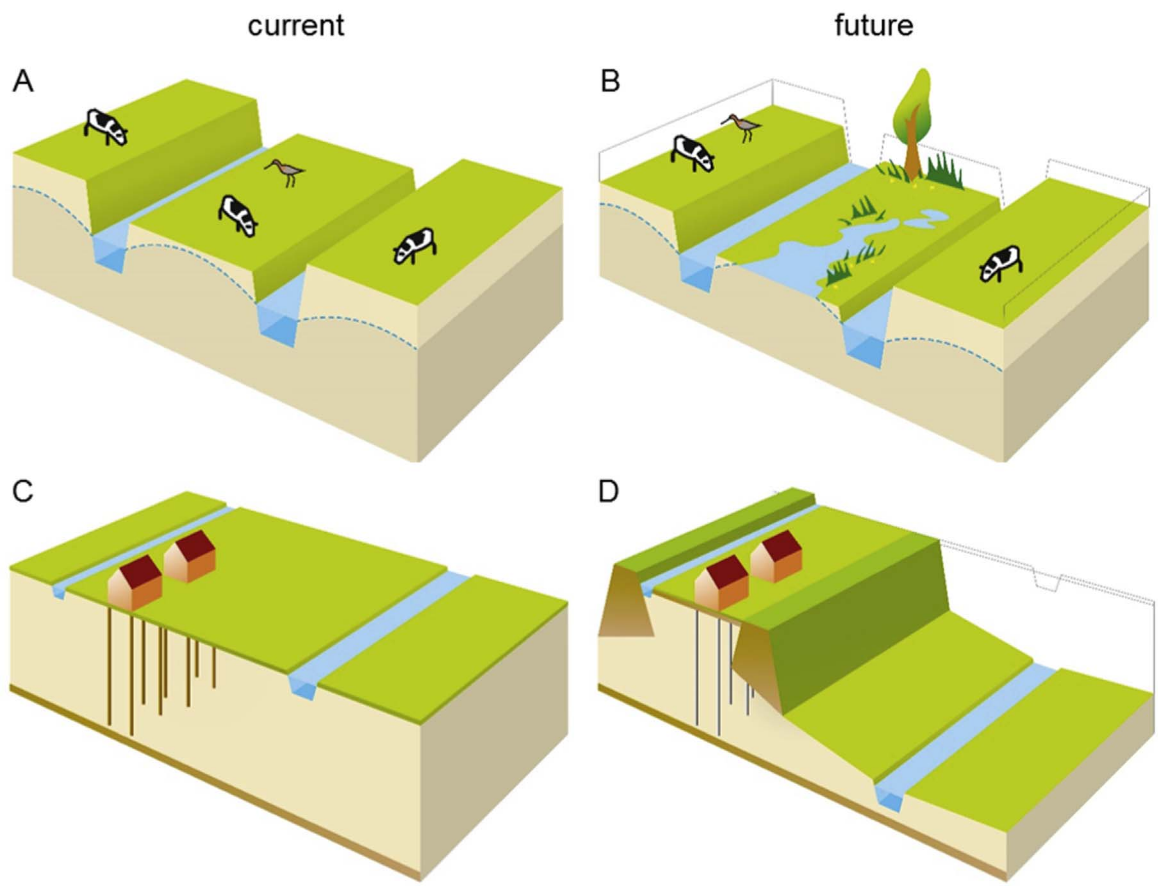

Fig. 3. Effects of spatial differences in soil subsidence in Dutch peatlands on depth of the groundwater table, and associated possible land use (A and B), and need for additional embankments, to prevent real estate damage (C and D).

$$\Delta S = 23.54 * ADG - 18.34 * CL - 6.68 \quad (1)$$

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 > 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 strategies 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).

2.4. 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.

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, 1525 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. 3C and D). The surface water levels in these watercourses must be kept high, because the house foundations (vertical lines underneath the houses in Fig. 3C and D) require high groundwater tables to prevent damage. In contrast, surface water levels in adjacent agricultural fields (right hand side of Fig. 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. 3D).

The GIS-model for weirs and embankments (model 2 in Fig. 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 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.

The GIS-model for real estate damage (model 3 in Fig. 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 1). Except for concrete foundation poles, all foundation types are prone to damage if ADG falls below the thresholds mentioned in Table 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 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.

Table 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	1730	Shallow brickworks	50
1920–1959	1670	Wooden poles	20
1960–1974	2933	Wooden poles topped with 50 cm concrete	70
1975–1989	3926	Wooden poles topped with 100 cm concrete	120
> 1990	4007	Concrete poles	None

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. 3A and B). The most influential parameter for agricultural yield is the groundwater table (dotted lines in Fig. 3A and B). If the groundwater table becomes too shallow crop yield diminishes (depicted by meadow birds that prefer high groundwater tables in Fig. 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. 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) 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 Brouwer and Spaninks (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. > 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). 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 reflects 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.

3. Results

3.1. Water levels and soil subsidence

The spatial differences in soil subsidence are pronounced (Fig. 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 > 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.

3.2. Societal impacts

The societal impacts show distinguished temporal trends (Fig. 5) and marked spatial patterns (Fig. 6). The maintenance of roads, sewers, and utility infrastructure linearly reflects the cumulative soil subsidence (Fig. 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. 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. 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. 5H). The reason is that almost all damage relates to houses built before 1960. The damage threshold for these houses (see Table 1) is most frequently breached in the first half of the 21st century.

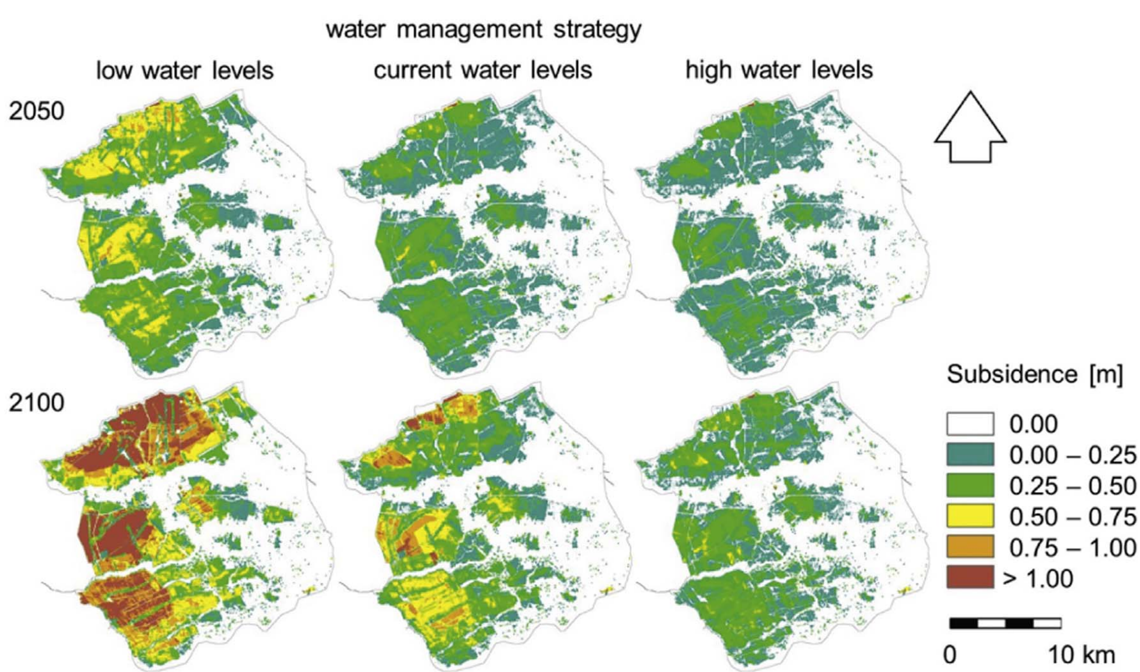

Fig. 4. Cumulative soil subsidence since 2010 in the years 2050 and 2100 due to the three water management strategies.

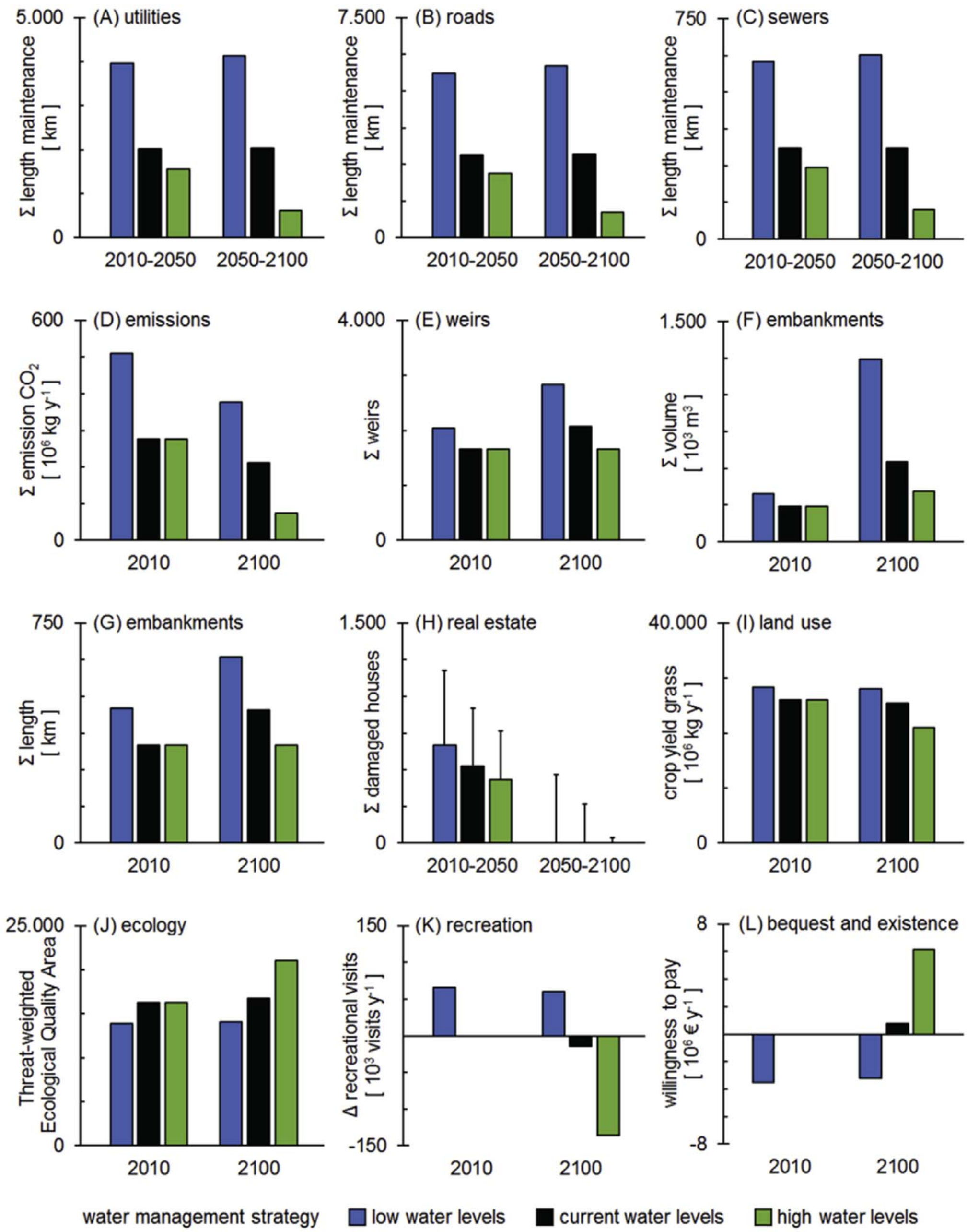

Fig. 5. Societal impacts of the three water management strategies. The error bars (H) reflect the range in age expectancy of real estate.

The magnitude of the downward trend is uncertain because it is unclear what the lifespan of the present real estate will be. At present, $< 5\%$ 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.

Land use is clearly dependent on the water management and the resulting soil subsidence (Fig. 6). In areas with limited soil subsidence (see Fig. 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. $> 20\%$ 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. 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. 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.

T-EQA values (Fig. 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. 5L) reveal a similar pattern. The impacts on recreational visits

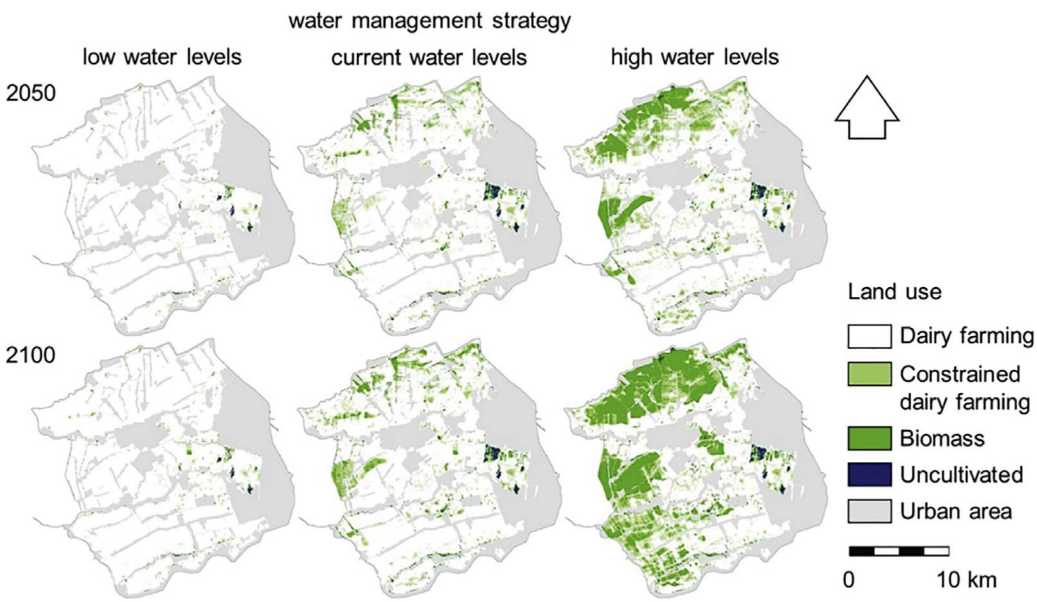

Fig. 6. Predicted land use in the years 2050 and 2100 due to the three water management strategies.

(Fig. 5K) are opposite, because opportunities for large-scale recreational activities such as fairs decrease as ecological values increase.

A comparison of the impacts of the management strategies reveals that they affect the stakeholders unequally (Table 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.

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. 3D). Our research raised awareness that continuation of this policy in the long run would require many additional embankments (Fig. 5G), while simultaneously the prevented real estate damage diminishes (Fig. 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 decreases 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

Table 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

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). From this we may conclude that our modelling 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).

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.

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Appendix A. Limitations of the soil subsidence assessment

A.1. Linear soil subsidence equation

The reliability of the soil subsidence model is limited by the linear character of the soil subsidence relation in Eq. (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., 2010). 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. Eq. (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, Eq. (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.

A.2. Empiric equation for current Dutch climate conditions

Moreover, Eq. (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 Eq. (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 Eq. (1) throughout time, and (2) changes in the ADG throughout time.

The empirical ADG-constant in Eq. (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 Eq. (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 A1). 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 A1
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 A1). 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.

A.3. No compaction

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.

A.4. Uncertain empiric data

Eq. (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 A1). 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.

A.5. 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) 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 Eq. (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 > 1 m, 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.

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

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 2.3). 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 Eqs. (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.

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

$s(T,h)$ = 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 (−)

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

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)

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.

Hoogland et al. (2012) used Eqs. (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.

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

$dE(s,t)$ / 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})

$$\begin{aligned} fO(s,t) &= P(s,t)/D \text{ if } P(s,t) \le D \\ fO(s,t) &= 1, \text{ if } P(s,t) > D \end{aligned} \tag{A5}$$

$P(s,t)$ = 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)

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 2.3). However, our research area is > 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.

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