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# Production in peatlands: Comparing ecosystem services of different land use options following conventional farming

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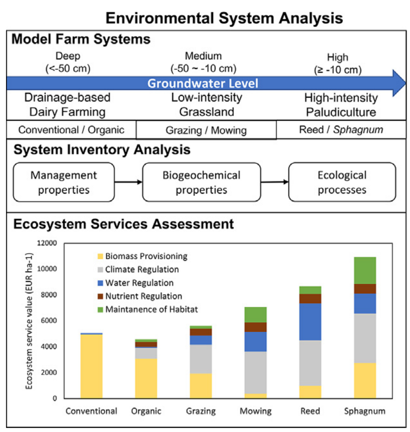
## HIGHLIGHTS

- Six theoretical model farm systems were defined for land uses under a gradient of water level.
- High water level leads to co-benefits of regulation and maintenance services.
- Market value of paludiculture production is not comparable with conventional dairy production.
- Sustainable peatland use needs fundamental management changes and financial support.

## GRAPHICAL ABSTRACT

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## ABSTRACT

Majority of Dutch peatlands are drained and used intensively as grasslands for dairy farming. This delivers high productivity but causes severe damage to ecosystem services supply. Peatland rewetting is the best way to reverse the damage, but high water levels do not fit with intensive dairy production. Paludiculture, defined as crop production under wet conditions, provides viable land use alternatives. However, performance of paludiculture is rarely compared to drainage-based agriculture. Here, we compared the performances of six land use options on peatland following a gradient of low, medium, and high water levels, including conventional and organic drainage-based dairy farming, low-input grasslands for grazing and mowing, and high-input paludiculture with reed and *Sphagnum* cultivation. For each land use option, we conducted environmental system analysis on model farm system defined by a literature based inventory analysis. The analysis used five ecosystem services as indicators of environmental impacts with a functional unit of 1-ha peat soil. Ecosystem services included biomass provisioning, climate, water, and nutrient regulation, and maintenance of habitat. Results showed that drainage-based dairy farming systems support high provisioning services but low regulation and maintenance services. Organic farming provides higher climate and nutrient regulation services than its conventional counterpart, but limited overall improvement due to the persistent drainage. Low-intensity grassland and paludiculture systems have high regulation and maintenance services value, but do not supply biomass provisioning comparable to the drainage-based systems. Without capitalizing the co-benefits of

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regulation and maintenance services, and accounting for the societal costs from ecosystem disservices including greenhouse gas emission and nitrogen pollution, it is not likely that the farmers will be incentivized to change the current farming system towards the wetter alternatives. Sustainable use of peatlands urges fundamental changes in land and water management along with the financial and policy support required.

## 1. Introduction

Peatlands are valuable ecosystems providing a wide range of goods and services to human society. Pristine peatlands serve as the largest natural terrestrial carbon store with global total carbon pool of over 600 Gt C (Leifeld et al., 2019; Yu et al., 2010), therefore provide vital climate regulation services. Hydrological functions of peatlands provide water related ecosystem services such as fresh water provisioning, flood control and mitigation of drought (Joosten and Clarke, 2002; Parish et al., 2008). The water-energy-nutrient dynamics in peatland ecosystems supports the maintenance of population and habitat for a large variety of specialized endemic species (Minayeva et al., 2017; Parish et al., 2008). These important ecosystem services related to carbon, water, nutrient and biodiversity are characterized and supported by the unique function of natural peatlands that features high water level and moisture content (Minayeva et al., 2017).

Peatlands also provide substantial provisioning services of biomass through plant cultivation and animal rearing for nutrition, materials and energy. However, exploitation of these provisioning services through intensive land uses have caused severe degradation and damages to the above-mentioned ecosystem services. Up to 25 % of global peatlands are degraded, yet they contribute to over 1.9 Gt $CO_2$ emissions annually (UNEP, 2022). Western Europe has the highest human impact on peatlands, where the majority of temperate peatlands are historically drained or currently under drainage for agricultural uses (Joosten and Clarke, 2002; UNEP, 2022). Western European countries including the Netherlands, Germany, UK and Ireland are among the highest in wetland losses in the world, losing over 70 % of wetland areas over the past three centuries (Fluet-Chouinard et al., 2023). The current state of intensive peatland uses has led to various negative environmental impacts. For example, intensively drained peat meadows are a substantial source of greenhouse gas (GHG) emissions (Schrier-Uijl et al., 2014; Tiemeyer et al., 2016). In the Netherlands, peatlands make up 15 % of the total agricultural land but (disproportionally) emit 35 % of all GHG emissions from the agricultural sector (Greifswald Mire Centre, 2020), which is associated with soil subsidence and subsequently increased flood risk and damage to infrastructure (Erkens et al., 2016). Decomposition of peat soil also contributes to the discharge of nutrients into surface water, together with leaching from intensive agriculture, adding to the deteriorating water quality and eutrophication (van Beek et al., 2007).

Water level is an important driver of the provisioning of ecosystem services in peatlands. Rewetting peatland to its natural hydrology is the best way to mitigate or even reverse the degradation, leading to long-term co-benefits of GHG emissions reduction (Günther et al., 2020), flood and drought mitigation (Ahmad et al., 2021, 2020), and biodiversity restoration (Strobl et al., 2020; Tuittila et al., 2000). However, current agricultural land uses are not suitable under high water levels (Tanneberger et al., 2021). The large area of agriculturally used peatlands cannot be taken out of production and rewetted solely on nature restoration purposes while ignoring the livelihood of the farmers. Nonetheless, evidence from peatland ecological monitoring studies proved that the climate benefit from raising water level in agricultural peatlands can be achieved without necessarily halting the productive use (Evans et al., 2021). 'Paludiculture', defined as productive land use of peatlands under rewetted condition (Wichtmann et al., 2016), presents a wide range of alternative production options at a gradient of higher water levels (Tanneberger et al., 2021) that could achieve these benefits. A variety of wet crops were identified, with management guidelines developed and farm-level feasibility evaluated (Geurts et al., 2019; Wichmann, 2017; Wichmann et al., 2020). On the other hand, agricultural sciences put great effort on the optimization of conventional dairy production systems which would lead to the continuation of the intensive drainage. Research on improving sustainability of dairy farming often focuses on organic farming practices with regard to optimizing feed and manure management (Baldini et al., 2018; Van Middelaar et al., 2014), but disregards peatland degradation and associated effects on soil carbon and nutrient dynamics, water, and biodiversity.

Integration of existing knowledge on peatland uses from both ecological and agricultural perspectives is urgently needed to support a transition to sustainable peatland use. Holistic evaluation of the performances of the land use options is required to understand their feasibility from a farmer's perspective, therefore support policy and decision making to incentivize land use transition. However, direct comparison of land use options is difficult despite the wide variety of field-based research available on peatland system. On the one hand, paired comparison is difficult given the field experiments and pilot studies are carried out at different locations, under different groundwater fluctuation and management regimes, and at different time scales. On the other hand, setting up new field experiment to make direct comparisons is time and money consuming, as it takes at least two to three years of continuous monitoring, after the vegetation succession has stabilized, before the effects of peatland rewetting projects can be detected, (Günther et al., 2017). As a result, drainage-based agricultural peatlands are rarely compared with paludiculture alternatives. Meanwhile, hypothetical system analysis has the potential to provide evidence on the environmental impacts of different systems without need of sophisticated experimental design and time and money investments. For example, a recent study from de Jong et al. (2021) compared emissions and revenues on a 1 ha unit of Dutch peat soil used for dairy production and cattail paludiculture. Such analyses can provide insights on the differences between systems and the causes by adopting field-based data when necessary, without need of a site-specific 'true-value'. However, comparisons covering multiple drainage-based agricultural land uses and paludiculture options are not yet available.

This article aimed to address the above-mentioned knowledge gap by providing a holistic view of the environmental impact of common productive land use options on peatlands following a gradient of groundwater levels. To achieve this goal, we selected six land use options under a gradient of groundwater levels, including drainage-based dairy farming under conventional and organic management, low-intensity grasslands for grazing and mowing at medium water levels, and high-intensity paludiculture with reed and *Sphagnum* cultivation at high water levels. For each land use option, we defined conceptual model farm system together with its farm structure and biogeochemical properties. We used an environmental system analysis (ESA) approach similar to the cradle-to-farm-gate partial life-cycle analysis (LCA). We evaluated five essential provisioning, regulation and maintenance ecosystem services as indicators of environmental impacts of the model farm systems.

## 2. Methods

### 2.1. Environmental system analysis

The performances of land use options on peatland were evaluated by an ESA using a method similar to the patrial LCA. LCA is an ESA tool to assess the potential environmental impacts and resources used throughout a product's life cycle from raw material acquisition, via production and use phases, to waste management (Finnveden et al., 2009). An LCA

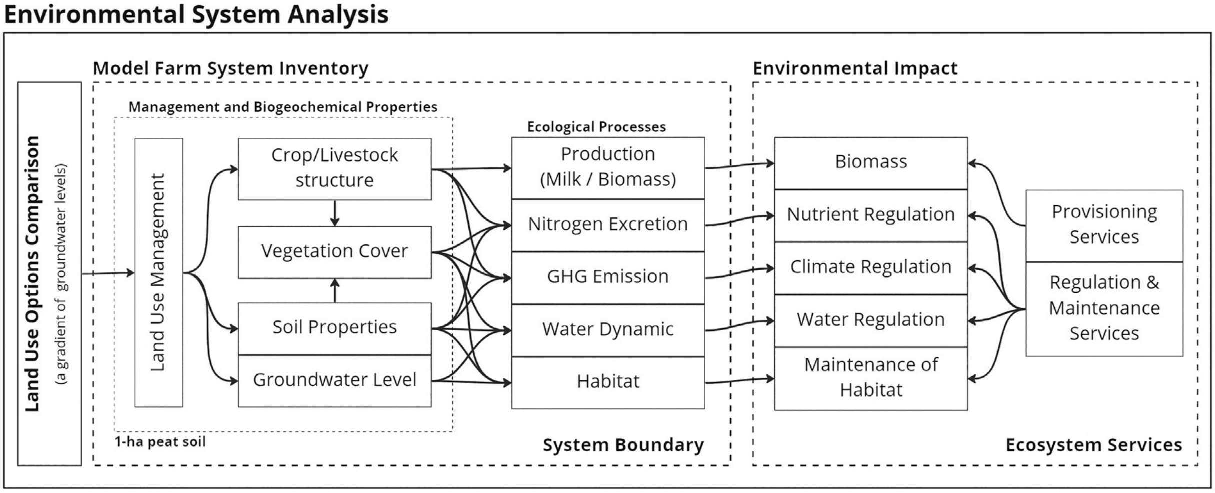


**Fig. 1.** Schematic overview of the ESA, from the goal of the analysis to the inventory of model farm systems, and linking to the environmental impact assessment using ecosystem services as indications.

study consists of four phases: goal and scope definition, inventory analysis, impact assessment, and interpretation (Finnveden et al., 2009). In order to provide a holistic view of productive land use options on peatlands, our ESA deviated from the standard LCA approach with different goals and scope defined below (Fig. 1).

We selected six land use options on peat soils following a gradient of managed groundwater level ranges as the scope of the analysis. Three groundwater categories were defined based on a summary of paludiculture options represented under different ranges of annual mean water level from Tanneberger et al. (2021): 50 cm below surface or dryer supports conventional drainage-based dairy farming systems. 50 to 10 cm below surface supports low-input production on grasslands where biomass from spontaneously established vegetation is harvested. High water level around or above surface level supports high-input paludiculture of deliberately established and selected wetland crops. Each groundwater category was represented by two land use options with different management and product types. The model farm systems were defined by an inventory analysis integrating published data and relationships from empirical studies, statistics and demonstrative cases (Rotz et al., 2010; de Jong et al., 2021). A common cradle-to-farm-gate system boundary was assumed, and a functional unit of 1-ha peat soil was defined for the model farm systems, which is in line with other environmental impact analysis of agricultural systems that reflects the impact of land use changes (e.g., Thomassen et al., 2008; Rotz et al., 2010; Baldini et al., 2018; de Jong et al., 2021).

The scope of the inventory analysis (Section 2.2), however, deviated from the LCA approach that focuses on input and output of energy and material under different land use management. We included peatland biogeochemical properties and ecological processes related to soil, water, and vegetation (Fig. 1). Only the processes occurring directly within the system boundary were included, e.g., on-site GHG emissions and pollutant excretion. To best represent the current state of land uses on temperate peatlands, we used statistical data from the Netherlands to define model systems under drainage-based dairy farming, and published data from case studies and pilot sites from western Europe to define model systems under paludiculture practices. We integrated state-of-the-art literature on peat ecological processes from peatland vegetation, ecohydrology, and soil biogeochemistry studies. The land use management, biogeochemical properties and ecological processes inventory was linked to ecosystem services (Fig. 1) as indicators for the model farm systems' environmental impact (Section 2.3). Following the CICES v5.1 classification system (Haines-Young and Potschin, 2018), we selected five essential ecosystem services including biomass under the provisioning services section, climate, water, nutrient regulations and maintenance of habitat under the regulation and maintenance services section. Both biogeochemical indicators and monetary values were assessed.

### 2.2. Model farm system inventory

The model farm system inventory consisted of three parts. Table 1 shows the land use management and associated biogeochemical properties. Land use management was characterized by the groundwater level range and the production types (Tanneberger et al., 2021). Management and biogeochemical specifications of the model farm systems under the defined land use options were then specified, including groundwater level, crop and livestock structure, vegetation cover, and resource (manure and

**Table 1**
Model farm system inventory of management and biogeochemical properties, including groundwater level, vegetation cover, crop and livestock production, and resource consumption.

| Assumption | Drainage-based dairy farming | | Low-intensity grassland | | High-intensity paludiculture | |
|---|---|---|---|---|---|---|
| | Conventional | Organic | Grazing | Mowing | Reed | Sphagnum |
| Annual mean water level (cm above surface) | −50 | −50 | −30 | −20 | 20 | −10 |
| Vegetation cover (main crop) | Species poor grassland (perennial ryegrass) | | Grazed nutrient-rich grassland (mixed fodder with perennial ryegrass) | Marshy hay meadow (mixed fodder with reed canary grass) | Reed (*Phragmites australis*) | Mire vegetation (*Sphagnum* mosses) |
| Stocking density (dairy cattle/young stock $ha^{-1}$) | 1.9/0.6 | 1.0/0.3 | 0.6/0.2 | 0 | 0 | 0 |
| Share of grassland (%) | 87 | 93 | 100 | 100 | 0 | 0 |
| Artificial fertilizer (kg N $ha^{-1}$) | 26 | 0 | 0 | 0 | 0 | 0 |
| Manure application (kg N $ha^{-1}$) | 254 | 169 | 0 | 0 | 0 | 0 |
| Diesel consumption (L $ha^{-1}$) | 104 | 96 | 57 | 64 | 36 | 229 |

artificial fertilizer, energy) consumption. Data was aggregated from various sources including national statistical data (e.g., Wageningen Economic Research, 2022) and empirical literature (e.g., Wichmann et al., 2020) for western Europe. Table 2 shows the ecological processes inventory. Primary and secondary production was determined according to the crop and livestock structures. Other ecological processes regarding carbon and nitrogen cycling, water dynamic, and biodiversity were characterized mainly by peatland vegetation-soil-water interactions. Specifically, for each model farm system, a reference vegetation type was selected from the Dutch vegetation classification system (Schaminee et al., 1995) matching the defined groundwater level and crop type. Bioindication functions of vegetation was used to quantify carbon and nitrogen emission (Tables S1, S3) factors (Couwenberg et al., 2011; Joosten et al., 2015; Liu et al., 2020b), groundwater dynamic (Table S2) (Everts and de Vries, 1991; Schaminée et al., 2012), and habitat quality (Table S4). Details of the inventory of the model farm systems are described below.

### 2.2.1. Drainage-based dairy farming – conventional/organic

The first system is the drainage-based dairy farming system. The Conventional system is the business-as-usual option in the Netherlands (de Jong et al., 2021; Joosten, 2010), and was proven to emit large amount of $CO_2$ through peat oxidation (Tiemeyer et al., 2020, 2016) and $CH_4$ from ruminant cows (Olesen et al., 2006). Management with low annual mean groundwater level of −50 cm and high nutrient input via artificial and manure fertilization was assumed to support high yield of perennial ryegrass (*Lolium perenne*). The resulting high grass yield of over 10 t DM $yr^{-1}$ (Weideveld et al., 2021) allows intensive dairy production with high stock density and milk productivity. This is the dominating crop for dairy-producing temperate peatlands due to a combination of management including controlled drainage, fertilization and re-sowing. Such vegetation cover was classified as species poor grassland due to less-productive species being outcompeted by ryegrass. A share of the land was characterized as arable land to produce maize as feed for the cows.

The second system Is the Organic'Iir' farming system. It was assumed to be free of artificial fertilization, with more grassland and less arable land, allowing more grazing hours for the cows. However, maintained intensive drainage and nutrient input through manure application sustained the dominance of perennial ryegrass, resulting in the same vegetation cover of species poor grassland as on the Conventional system. Grass and crop yield was assumed lower than in the Conventional system (Olesen et al., 2006; Thomassen et al., 2008). Lower number of cows and milk productivity was supported as a consequence.

For both conventional and organic dairy farming, detailed data on farm structure and field management was retrieved from an online database agrimatie (Wageningen Economic Research, 2022, www.agrimatie.nl) (Table S1). Young stock was assumed to be kept at 30 % of the stocking density to replace dairy herd (van Boxmeer et al., 2021). The amount of manure excretion was adopted from the Dutch national statistics per milk productivity category (RVO, 2021a), and assuming all manure being applied on the field. Artificial fertilizer was added to the conventional dairy farm to match the maximum allowed nitrogen input at 300 kg N $ha^{-1}$ by Dutch regulation (RVO, 2021b). Fertilizer application in Table 1 was presented as average values per hectare. Diesel consumption on farm was adopted from Thomassen et al. (2008).

### 2.2.2. Low-intensity grassland – grazing/mowing

Low-intensity grassland systems involve two different biomass uses on grasslands under extensive management. One is a grazing system that aims at producing milk, the other is a mowing system that produces biomass as bioenergy feedstock for direct combustion. Both are not yet widely applied as farming practices, but sufficient case studies in western Europe exist (Wichtmann et al., 2016) to base the assumptions on. Dairy milk was designed as product for the Grazing system for direct comparison to the intensively drained systems. Other forms of bioenergy (e.g., bioethanol) was not considered due to the lack of field-based data source and the extra complication of parameterization it will introduce. Both systems did not have the intensive water management required to maintain deep or very high water levels. Input of extra nutrients was also absent. Grazing use of the grasslands was assumed under shallow-drained conditions with annual mean water level at around −30 cm. Examples of intensive cattle-grazing systems at this water level can be found in the Netherlands (Weideveld et al., 2021), or at even higher water level (ca. −21 cm) in Germany (Poyda et al., 2017). Vegetation of grazed nutrient rich grassland type fitted with such water and livestock management regimes. Large inter-annual water level fluctuation observed on such vegetation (Table S2) allows animal activities during dry seasons that covers large parts of the growing season, while high water levels during wet seasons facilitates the development of richer grass species. The presence of

**Table 2**
Model farm system inventory of ecological processes including habitat type, GHG emissions, nitrogen excretion, water dynamics, and agricultural production.

| | Drainage-based dairy farming | | Low-intensity grassland | | High-intensity paludiculture | |
|---|---|---|---|---|---|---|
| | Conventional | Organic | Grazing | Mowing | Reed | Sphagnum |
| Reference vegetation type | Species poor grassland (Poa tricialis-*Lolium perenne*) | | Grazed nutrient-rich grassland (e.g. Lolio-Cynosurion) | Marshy hay meadow (e.g. Calthion palustris) | Reed (Phragmition australis) | Mire vegetation (e.g. Scheuchzerietea) |
| GEST type[1] | G1/A1 Dry to moderately moist grassland/arable land | G1/A1 Dry to moderately moist grassland/arable land | G2 Moist grassland | G3s Moist to very moist grassland with shunt species | U15 Very wet Phragmites and Phalaris reeds | U13 Wet *Sphagnum* lawn |
| $CO_2$ Emission factor (t $ha^{-1}$ $yr^{-1}$) | 22.07 | 21.53 | 12.95 | 14.1 | −7.23 | −3.44 |
| $CH_4$ Emission factor (t $CO_2$-eq $ha^{-1}$ $yr^{-1}$) | 0.002 | −0.02 | 0.01 | 0.75 | 11.97 | 3.10 |
| $N_2O$ Emission factor (t $CO_2$-eq $ha^{-1}$ $yr^{-1}$) | 2.19 | 2.19 | 0.75 | 0.005 | 0.005 | 0.005 |
| Manure excretion (kg N per dairy cattle/young stock) | 124/76 | 104/76 | 104/76 | 0 | 0 | 0 |
| Annual maximum water level (cm above surface) | −20[2] | −20[2] | 2[3] | 5[3] | 30[4] | 5[3] |
| Flood tolerance (days) | 14[5] | 14[5] | 14[5] | 49[6] | Permanent (365 days per year)[4] | 105 (3.5 months)[7] |
| Yield ($ha^{-1}$) | 13.5 t Milk | 6.2 t Milk | 3.9 t Milk | 6 t DM | 500 bundle | 110.3 $m^3$ |

[1] van Belle and Elferink (2020), Couwenberg et al. (2011), Couwenberg et al. (in prep). Emission factors calculated based on the share of grassland (Table 1) and excluding CO2-eq of harvested grass biomass where applicable (Table S1); [2] Hennekens et al. (2010); [3] Everts and de Vries (1991); [4] Geurts and Fritz (2018); [5] McFarlane et al. (2003); [6] Wrobel et al. (2009); [7] Rochefort et al. (2002).

ryegrass (*Lolium perenne)* provides high quality fodder, together with bentgrass (*Agrostis* spp.) and fescues (*Festuca* spp.) species with lower fodder quality to supply feed for dairy cows. The overall lower energy content of the fodder led to the assumption of a lower number of cows, while the milk productivity was assumed to be at the same level with the Organic system according to the case study of Bakker and Ter Heerdt (2005). No fertilization was allowed, with only manure excretion as nitrogen input also at the same amount as the Organic system. Diesel consumption was assumed to be proportional to Organic based on animal density.

Without needs for animal production, higher water level at −20 cm was defined on the Mowing system, leading to a vegetation cover of marshy hay meadows. Mixed folder is produced with reed canary grass dominance and presence of sedges (*Carex* spp.), rushes (*Juncus* spp.) and marsh-marigold (*Caltha palustris*). The presence of rushes was frequently observed on rewetted agricultural peatlands (Lamers et al., 2015), which are 'shunt species' that have aerenchyma that would provide an extra pathway for $CH_4$ effluxes (Couwenberg and Fritz, 2012). No grazing animal or fertilization was defined for this system. Biomass yield of 6 t DM ha$^{-1}$ was adopted from case studies of biomass production on German peatlands (Wichtmann et al., 2016). Diesel consumption was calculated based on the average fuel consumption of mowing and transporting in L h$^{-1}$ and the operation time in h ha$^{-1}$ (Wichmann, 2017).

#### 2.2.3. High-intensity paludiculture – reed/*Sphagnum*

High-intensity paludiculture systems aimed to produce wetland crops since the very high groundwater level do not allow animal related production. Reed and *Sphagnum* mosses were selected as representative crops as they have been proven viable according to pilot studies (Geurts and Fritz, 2018; Wichmann et al., 2020), and the products have demonstrated economic potential (Müller and Glatzel, 2021; Wichmann, 2017). The reed system requires high water level at 20 cm above surface for best crop performance (Geurts and Fritz, 2018, Table S2). The most profitable product was selected, which is reed bundles for roof thatching at a yield of 500 bundles ha$^{-1}$ (approximately 8 t DM ha$^{-1}$) (Wichmann, 2017). Growth of peat mosses in the *Sphagnum* system requires water levels close to surface at an annual mean of −10 cm. Management of the *Sphagnum* system was characterized by harvest once in five years with annual average yield of 110.3 m$^3$ ha$^{-1}$ yr$^{-1}$ in volume (approximately 3.2 t DM ha$^{-1}$ yr$^{-1}$) as horticulture growing media (Wichmann et al., 2020). No extra nutrient input was defined for both systems. However, the reed system needs higher nutrient levels from soil or surface water to support the high biomass yield (Geurts et al., 2020). The *Sphagnum* system requires removal of the nutrient-rich topsoil for the establish of *Sphagnum* mosses (Huth et al., 2021). Diesel consumption was calculated and divided into annual averages based on the time and fuel consumption factors for harvesting and on-farm transportation provided by Wichmann (2017) and Wichmann et al. (2020).

### 2.3. Ecosystem services assessment

#### 2.3.1. Climate regulation

Climate regulation service was indicated by the reduction of GHG emissions. Direct gaseous emission sources within the farm boundary were quantified using emission factors and farm management properties from the model farm inventory. Emission sources were divided into field emission and farmyard emission. All GHGs are converted into $CO_2$ equivalent ($CO_2$-eq) using global warming potential (GWP) under 100-year time horizon (27.0 for CH4 and 273.0 for N2O, Forster et al., 2021). Reduction of emissions was determined by comparing all model farm systems to the conventional dairy farm as a baseline. Detailed parameter values and calculations are presented in Supplement Table S1.

*2.3.1.1. Field emissions.* Field emission covered all the GHG emissions from the plant–soil interface, which includes $CO_2$ fluxes from net ecosystem exchange (NEE) (Veenendaal et al., 2007), $CH_4$ effluxes via diffusion and ebullition (Couwenberg and Fritz, 2012), and direct $N_2O$ emission from peat soil mineralization. The most important driving factors are groundwater level and land cover (Tiemeyer et al., 2020). Peatland field emissions were rarely allocated into agricultural system analysis, but necessary for a complete representation of the environmental impacts from productive peatlands. Earlier studies have included soil carbon change into life cycle assessments of dairy farms on mineral soils, which resulted in lower GHG emissions due to carbon sequestration (Knudsen et al., 2019; Salvador et al., 2017). However, direct measurements of carbon balance on drained peat soils often reported strong carbon sources (Schrier-Uijl et al., 2014; Weideveld et al., 2021). Therefore, peatland-specific methodology should be incorporated into this ESA. Currently, the best estimate available for peat soil emissions at project level is the Greenhouse Gas Emission Set Type (GEST) approach (Ekardt et al., 2020). This approach provides $CO_2$ and $CH_4$ emission factors to vegetation types based on meta-analysis of yearly fluxes measurements in relation to groundwater level and vegetation cover (Couwenberg et al., 2011; van Belle and Elferink, 2020; Couwenberg et al., in prep). GEST types were selected based on the vegetation cover assumed for each model farm system (Table 1) and the corresponding emission factors were assigned (Table 2). For dairy producing systems, carbon in harvested biomass was converted into $CO_2$-eq and subtracted to avoid double accounting with livestock-related emissions, because it was already included in the GEST $CO_2$ emission factors of grassland vegetations (Couwenberg et al., 2011). A carbon content of 42.5 % and a total loss of 27 % during management was applied on the dry matter yield of grass biomass (van Schooten and Philipsen, 2012). Field $N_2O$ emission factor was adopted from the Dutch national inventory (van der Zee et al., 2021) for drained peatlands, from the IPCC Tier 1 emission factors (IPCC, 2013) for the shallow-drained grasslands and from a meta-analysis (Tiemeyer et al., 2020) for rewetted peatlands.

*2.3.1.2. Farmyard emissions.* Farmyard emission covered all the direct emissions from production-related sources including animal keeping, manure management, and energy use, which were determined by the inventory of management properties. $CO_2$ from animal respiration per cattle was assumed to be 4.6 kg $CO_2$-C d$^{-1}$ (Felber et al., 2016). $CH_4$ and $N_2O$ emission factors were taken from the Dutch national inventory (van der Zee et al., 2021; Ruyssenaars et al., 2020). Annual $CH_4$ emission factors for enteric fermentation were 134.6 and 34.2 kg $CH_4$ cattle$^{-1}$ yr$^{-1}$ for dairy and young cattle. Annual emission factors related to manure management were 38.8 and 7.85 kg $CH_4$ cattle$^{-1}$ yr$^{-1}$ for dairy and young cattle; and 0.002 kg $N_2O$ kg$^{-1}$ nitrogen excretion. $N_2O$ emissions from fertilize use are 0.005 and 0.013 kg $N_2O$-N kg$^{-1}$ applied-nitrogen from manure and inorganic nitrogen fertilizers, respectively. Emissions from energy uses only included onsite emissions from direct fuel consumption. Off-site and indirect emissions from electricity generation, transportation, and processing were not considered. Emission factor of diesel was 3.23 kg $CO_2$-eq. L$^{-1}$.

*2.3.1.3. Monetary value.* GHG emission reduction was accounted as carbon credit. A carbon price of EUR 75 t$^{-1}$ $CO_2$-eq was adopted from the Dutch Green Deal National Carbon Market (www.nationaleco2markt.nl), where carbon credits from emission reduction in peat meadow areas were successfully sold.

#### 2.3.2. Water regulation

Water regulation service was indicated by the potential water storage of the model farm systems. Water could be stored both below surface as soil pore water and above ground when the system is inundated, with the risk of damaging the productivity of the crops. Therefore, volume of potential water storage was quantified based on groundwater level dynamics as below and above ground storages and supplemented by information on the flood tolerance of the vegetation. Detailed data sources and calculation are presented in supplement Table S2.

*2.3.2.1. Below surface water storage.* Efficiency of below surface water storage was indicated by soil specific yield, which measures the differences between total porosity and volumetric water content. It was calculated according to its correlation with peat bulk density (Liu et al., 2020a):

$$S_Y = 0.003 \times BD^{-1.4}$$

where $S_Y$ is soil specific yield, *BD* is bulk density. Bulk density of the peat soil was adopted from Lennartz and Liu (2019) and Liu and Lennartz (2019) according to its correlation to the land use type and the level of degradation. Intensive agricultural peat soil under deep drainage from the drainage-based dairy systems was assumed to be highly degraded with high bulk density (Weideveld et al., 2021) at 0.6 g cm$^{-3}$. Higher water level in the low-intensity grassland systems was assumed to improve soil conditions and lower bulk density (Ahmad et al., 2020), therefore assumed at 0.3 g cm$^{-3}$. With peat-forming vegetation cover of reed and *Sphagnum* mosses, high-intensity paludiculture systems were assumed to have the lowest bulk density at 0.2 g cm$^{-3}$ (Lennartz and Liu, 2019; Liu and Lennartz, 2019). Annual mean water level (Table 1) represented the volume of subsurface space that could be used for soil water storage. The total below surface water storage potential was therefore calculated as

$$S_Y \times WL_{mean} \times 1\ ha$$

where $S_Y$ is soil specific yield in cm$^3$ water cm$^{-3}$ soil, $WL_{mean}$ is annual mean water level in cm below surface.

*2.3.2.2. Above surface water storage.* Above surface water storage was indicated by both the amount of water and the duration of inundation that the vegetation can tolerant. The amount of water was calculated as:

$$\Delta WL_{max} \times 1ha$$

where $\Delta WL_{max}$ is the difference between maximum to mean water level above surface. Maximum water level was indicated by the vegetation cover of the systems according to a meta-analysis of past measurement records (Everts and de Vries, 1991) and a vegetation bioindication database for the Netherlands (Hennekens et al., 2010; Schaminée et al., 2012) (Tables 2, S2). A discount rate of 5 % was applied to the above surface water storage of the reed system, based on the diameter and density of reed shoots retrieved from Boar et al. (1999). The same rate was used for grassland systems, which will lead to a conservative estimate due to smaller standing biomass of grass comparing to reed. No discount was applied the *Sphagnum* system due to a combination of high water holding capacity of the moss layer and the thinness of vascular plants on top of it (Schouwenaars and Gosen, 2007). In addition, duration of inundation was represented by the flood tolerance of the main crop of the production system retrieved from literature, which indicates the vulnerability of the crops against inundation during growing seasons (McFarlane et al., 2003; Rochefort et al., 2002; Wrobel et al., 2009).

*2.3.2.3. Monetary value.* Monetary value of water storage was accounted by an avoided damage approach. Under a hypothetical heavy precipitation event that happens once per year, water stored in the production system was assumed to be able to avoid overflow that would cause damages with a price of EUR 3 m$^{-3}$ water, according to an analysis from the Dutch water authority (Kanters et al., 2016).

### 2.3.3. Nutrient regulation

Similar to the assessment of climate regulation service, nutrient regulation service was indicated by the reduction of nitrogen pollution. Nitrogen pollution was calculated based on emission factors mainly from the Dutch national inventory (van der Zee et al., 2021) and supplemented by empirical studies. Nitrogen pollution was divided into gaseous pollution, including $NH_3$ and $NO_x$ gases, and water pollution as nitrogen leaching. Detailed parameter values and calculations were presented in Supplement Table S3. $NH_3$ and $NO_x$ emissions from manure management were quantified based on the total ammonia nitrogen (TAN) as a fraction of total manure nitrogen excretion (Velthof et al., 2012). Proportions of the manure nitrogen excretion and manure application were emitted as $NH_3$ and $NO_x$ pollutants according to emission factors from the Dutch national inventory (van der Zee et al., 2021). Amount of nitrogen leaching was adopted from the Moorfuture® methodology (Joosten et al., 2015), where a nitrogen leaching factor in kg N ha$^{-1}$ yr$^{-1}$ was assigned to vegetation cover types, similar to the GEST approach, based on a meta-analysis for European temperate peatlands.

Monetary value of nutrient regulation service was quantified using an avoided cost approach. Marginal costs of different nitrogen pollutants were adopted from van Grinsven et al. (2013) (Table S3). Only the human health-related costs were included to avoid double accounting of the climate and biodiversity impacts that were included in the assessment of climate regulation service. Total cost of each model farm system was compared to the conventional drainage-based dairy farming system to represent the value of the reduction. To achieve a conservative estimation we used original monetary values provided by Van Grinsven et al. (2013) without inflation correction to 2020 market values.

### 2.3.4. Maintenance of habitat

Peatland habitats are highly variable and support different functional groups of the flora and fauna, making it impossible to be quantified with a simple numeric indicator even within a single taxa (Minayeva et al., 2017). Therefore, a qualitative approach was developed to assess the maintenance of habitat service using land use management practices as indications. Based on management properties of the model farm systems, commonly applied management techniques were identified, and their effects on peatland habitats qualitatively evaluated through literature review. Effect of the management techniques was summarized and determined as positive or negative, which supported a literal assessment of the habitat quality of all model farm systems, and simplified into a ranking score system representing system-level differences between the model farm systems (Table 3). Monetary value of the maintenance of habitat service was quantified using a willingness-to-pay approach (Farnsworth et al., 2015). Subsidy prices that the government was willing to pay for the six model farm systems were determined by matching vegetation cover types to the habitat types from the Dutch Subsidy system of Nature and Landscape (SNL, Subsidiestelsel Natuur en Landschap, www.bij12.nl) (Table S4). Ranking scores of the habitat quality were cross-validated with the monetary values to ensure the consistency of relative differences between the two indications of the model farm systems.

### 2.3.5. Biomass provisioning

The biomass provisioning service was indicated by the annual productivity data of food and raw materials from the model farm system inventory. Monetary value of the biomass service was indicated using direct revenue, multiplying productivity and market prices of the products. Use of revenue as an indication allowed direct comparison of biomass productivity among land use options with different final products, including milk, bioenergy, building material, and agricultural substrate (Tables 2, S5). The market price of milk was set at EUR 367.7 and 500.1 t$^{-1}$ Milk for conventional and organic systems (Wageningen Economic Research, 2022). Milk from the low-intensity grazing system was considered an organic product and assumed to have the same price as the organic system. The actual price can be variable due to the absent of established market and regulations for the low-intensity production system. The impact of this price on the biomass service assessment was assumed to be small, because of the substantially larger difference in the quantity of milk productivity between systems than differences between milk prices. Prices of biomass as bioenergy and building material were adopted from Wichmann (2017) at EUR 46 t$^{-1}$ dry matter for direct combustion in the low-intensity mowing system, and EUR 2 bundle$^{-1}$ of reed for roof thatching in the high-intensity reed system. The price of *Sphagnum* peat as growing media was EUR 25 m$^{-3}$ (Wichmann et al., 2020).

**Table 3**
Qualitative ranking scale of habitat quality of the model farm systems based on common management practices and their effects.

| Production system | Management (effect: −, negative; +, positive) | Habitat quality (Scale 1–5) | Subsidy type | Subsidy (euro $ha^{-1}$ $yr^{-1}$) |
|---|---|---|---|---|
| Conventional | Deep drainage (−); Tillage (−); Heavy fertilization (−); Intensive grazing/mowing (−) | Very low (1) | – | 0 |
| Organic | Deep drainage (−); Tillage (−); Manure application (−); Intensive grazing/mowing (−) | Low (2) | Open grassland (A11 Open grasland) | 197.93 |
| Grazing | Rewetting (+); No Tillage (+); Grazing manure (−); Extensive grazing (+) | Medium to high (3.5) | Flora- and fauna- rich grassland (N12.02 Kruiden- en faunarijk grasland) | 219.08 |
| Mowing | Rewetting (+); No Tillage (+); No fertilization (+); Extensive mowing (+) | High (4) | Moist meadow (N10.02 Vochtig hooiland) | 1188.42 |
| Reed | Rewetting (+); No Tillage (+); High nutrient load (−); Extensive harvest (+) | Moderate (3) | Mowed reedland (N05.02 Gemaaid rietland) | 575.98 |
| Sphagnum | Rewetting (+); No Tillage (+); No fertilization (+); Topsoil removal (+); Extensive mowing (+); Species re-introduction (+) | Very high (5) | Peat moss floating mat (N06.02 Trilveen) | 2082.26 |

### 2.4. Sensitivity analysis

Assumptions and limitations in data and methods of this ESA were considered as the main contributors of uncertainties in the results. Uncertainty of the parameter values, as results of measurement error, model accuracy, spatial and temporal variations from the literature adopted, was considered out of the scope of this analysis. A sensitivity analysis incorporating alternative methods of model farm system inventory analysis and environmental impact assessment was performed as a measure of uncertainty to evaluate possible bias in the ESA. Detailed data sources and calculations were presented in Supplement Table S6.

#### 2.4.1. Sensitivity of model farm system inventory

This analysis applied a common cradle-to-farm-gate system boundary. Model farm system inventory analysis was based on published relationships between soil, water, vegetation and management. Alternative methods regarding the definition of system boundary, choice of literature data and relationships, and use of indicator metrics were tested in this sensitivity analysis.

First, a cradle-to-grave system boundary (Finnveden et al., 2009) was analyzed in the GHG emissions inventory. $CO_2$-eq of carbon export through milk and biomass production was calculated, assuming all carbon in the final products would eventually become $CO_2$ emissions. Carbon export of milk was calculated assuming a carbon content of 12 g C $g^{-1}$ N (Rotz et al., 2010) and 30 %/70 % nitrogen secretion/excretion rates; for mowing and paludiculture systems carbon export was calculated using the carbon content of mixed fodder (45 %, Adamovics et al., 2018) and *Sphagnum* mosses (51 %, Roy et al., 2018).

Second, an alternative relationship between groundwater level and GHG emission factors was tested. Currently used emission factors adopted from the GEST approach was the best representation of the model farm system inventory in terms of management, groundwater, and vegetation properties. However, the number of data points characterizing each GEST type was limited due to the detailed vegetation classification (n = 4–24, Couwenberg et al., in prep). In this sensitivity analysis, different GHG emission factors were derived using a response curve of $CO_2$ and $CH_4$ emissions to annual mean water level (n > 140, Tiemeyer et al., 2020). Harvested carbon was also subtracted from the grassland systems to avoid double accounting. This approach represented a more generalized GHG emissions accounting method without representation of system-level characteristics.

At last, effect of changing the indication metric of GHG emission inventory from 100-year GWP to 20-year GWP was tested. Although the 100-year GWP is currently the most commonly used $CO_2$-eq metric, the weighting assigned to non-$CO_2$ short-lived GHGs (i.e., $CH_4$, $N_2O$) differs significantly depending on the time horizon of the metrics and will obscure major differences in the evaluation of climate impact (Lynch, 2019). Therefore, the effect of applying 20-year GWP, 79.7 for $CH_4$ and 273.0 for $N_2O$ (Forster et al., 2021), was assessed.

#### 2.4.2. Sensitivity of environmental impact assessment

In this analysis, environmental impact assessment using ecosystem services consisted of carbon credits potentially generated from avoided GHG emissions, avoided damages from reduced nitrogen pollution and flooding, subsidies for the maintenance of habitat, and market value of the products. These monetary values supported comparison of ecosystem services under an uniformed monetary unit, but could not represent the costs and benefits of the model farm systems. In order to shed light on the economic performance of the farm systems, another environmental impact assessment approach was tested in this sensitivity analysis, focusing on the operational cost of the model farm systems as well as the societal costs of ecosystem disservices. Rough estimates of management costs were adopted from literature despite the scarcity of available data and the subsequent uncertainties (Daatselaar and Prins, 2020; van de Riet et al., 2014; Sechi et al., 2022; Wichmann, 2017; Wichmann et al., 2020). Ecosystem disservices including GHG emissions and nitrogen pollutions were assessed in terms of potential societal costs of the damages, rather than being translated into climate and nutrient regulations services. GHG costs were represented by the proposed industrial carbon tax from the Dutch Climate Agreement (Klimaatakkoord.nl), that will reach EUR 125 $t^{-1}$ CO2-eq by 2030. Nitrogen damage costs were adopted from Van Grinsven et al. (2013). Assessments of water regulation and maintenance of habitats did not involve any direct quantification of ecosystem disservices, therefore not included in this analysis.

## 3. Results

### 3.1. Ecosystem services assessment

#### 3.1.1. Climate regulation

Overall, the climate regulation service of the model farm systems (Fig. 2a) was high for low-intensity grassland and high-intensity paludiculture systems, which had higher groundwater level and lower animal density and resource consumption (Table 1). More specifically, total GHG emissions (Fig. 2b) showed large differences between model farm systems following the groundwater level gradient. Differences of the field $CO_2$ emissions were especially in line with the groundwater level categories, where drainage-based systems emit large amount of $CO_2$, while the higher water level in low-intensity grassland systems led to less $CO_2$ emission. In high-intensity paludiculture systems where water level was assumed to be close to or above surface, negative $CO_2$ emissions was achieved with higher photosynthetic $CO_2$ uptake than emission from peat oxidation. However, large $CH_4$ emissions under such high water level led to net positive total emissions from the paludiculture systems. Field $N_2O$ emissions were only present in the dairy producing systems, while the rewetted grassland and high-intensity paludiculture systems emitted negligible $N_2O$. Farmyard emissions showed distinctive differences between dairy producing and biomass producing systems as well, mainly characterized by management properties, especially animal stocking density and fertilizer application (Table 1). Non-dairy producing

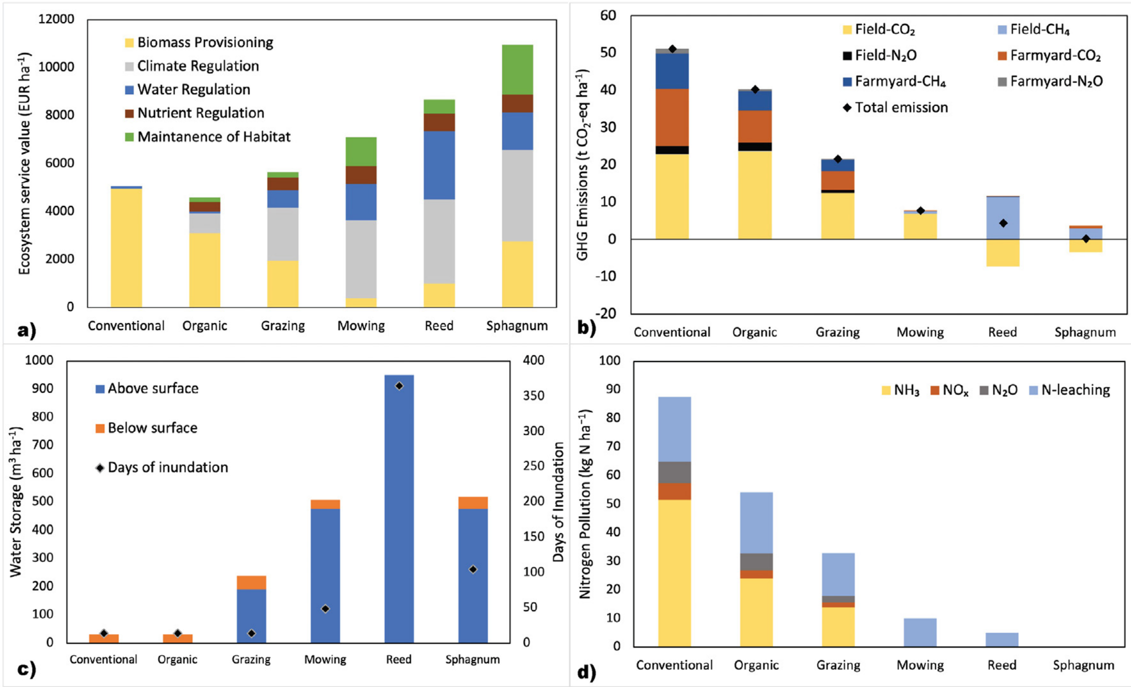


**Fig. 2.** Results of the ecosystem services assessment: a) monetary value of ecosystem services; b) GHG emissions per category; c) water storage potential per category; d) nitrogen pollution per pollutant type.

systems with energy consumption as the only source of field emission source showed substantially lower field emissions.

*3.1.2. Water regulation*

The value of water regulation service (Fig. 2a) was negligible on drainage-based dairy systems, and highest on the reed paludiculture system. Specifically, potential of water storage showed substantial differences between above and below surface and between different systems (Fig. 2c). Below surface water storages were limited comparing to above ground resulting from small soil water holding capacity on degraded peat soils of the drainage-based systems, and limited space between surface and groundwater level of the rewetted systems (Table 1). Above ground water storage was absent in the drainage-based dairy farm systems following a small range of groundwater dynamic; and was high on rewetted systems where the vegetation allowed higher maximum water level, especially on the reed system that was constantly under inundation (Table S2).

*3.1.3. Nutrient regulation*

Differences of the total nitrogen pollutions and the subsequent nutrient regulation service between the systems showed a similar pattern as GHG emissions (Fig. 2d) and the climate regulation service (Fig. 2a). Amount of total nitrogen pollution mainly followed the gradient of management properties including animal density and fertilizer application (Table 1). Ammonia ($NH_3$) emission was the largest source of nitrogen pollution in the dairy producing systems. Together with $NO_x$ and $N_2O$, livestock originated air-borne nitrogen pollution is substantially larger than the amount of nitrogen leaching into water bodies. Nitrogen leaching was high in all model farm systems, and was the sole pollutant for the non-dairy producing systems except the *Sphagnum* system, where nitrogen pollution was completely absent.

*3.1.4. Maintenance of habitat*

Value of the maintenance of habitat service (Fig. 2a) was absent in the conventional dairy farm system, low in the two other dairy producing systems – organic and grazing systems, and relatively high in the mowing grassland system and paludiculture systems. The management-based habitat quality ranking scale was in good agreement with the subsidy values of assigned vegetation and habitat types (Tables 3, S4). However, the maintenance of habitat service value was overall lower in comparison to other ecosystem services.

*3.1.5. Biomass provisioning*

Dairy production from the drainage-based dairy systems had the highest biomass provisioning services (Fig. 2a). Despite a higher market price, the organic dairy system yielded a lower revenue comparing to the conventional dairy system, due to lower animal density and per animal productivity (Tables 1, 2). Biomass provisioning service of the *Sphagnum* system is at the same level as the organic dairy system, characterized by a high market price of peat mosses while being limited by the low productivity of the system. Biomass from the low-intensity grasslands and Reed paludiculture system have substantially lower revenue, thus relatively low biomass provisioning service.

*3.2. Sensitivity analysis*

*3.2.1. Sensitivity of model farm system inventory*

Overall, changing data and assumptions in the inventory analysis did not alter the system-level difference between model farm systems (Fig. 3). First, inclusion of carbon export for a cradle-to-grave system boundary increased the net GHG emissions of the biomass-producing systems for over two times from the original analysis. However, GHG emissions were still higher in the dairy producing systems with major contributions from field and animal sources of GHG emissions.

Second, application of the water level response curve and the subsequently different emission factors did not change the dominant role of $CO_2$ in the GHG budgets of dairy producing systems. The total emission is roughly in agreement with the initial estimates except for the substantially higher $CO_2$ emissions in the low-intensity grassland systems with shallow water levels. $CH_4$ from the reed system was higher than the original estimate. $N_2O$ estimates are in agreement with the formal analysis and remain low across systems. In summary, alternative choice of data and relationships gave more conservative estimates for GHG emissions, i.e., higher emissions and lower carbon uptake, but did not affect the high climate

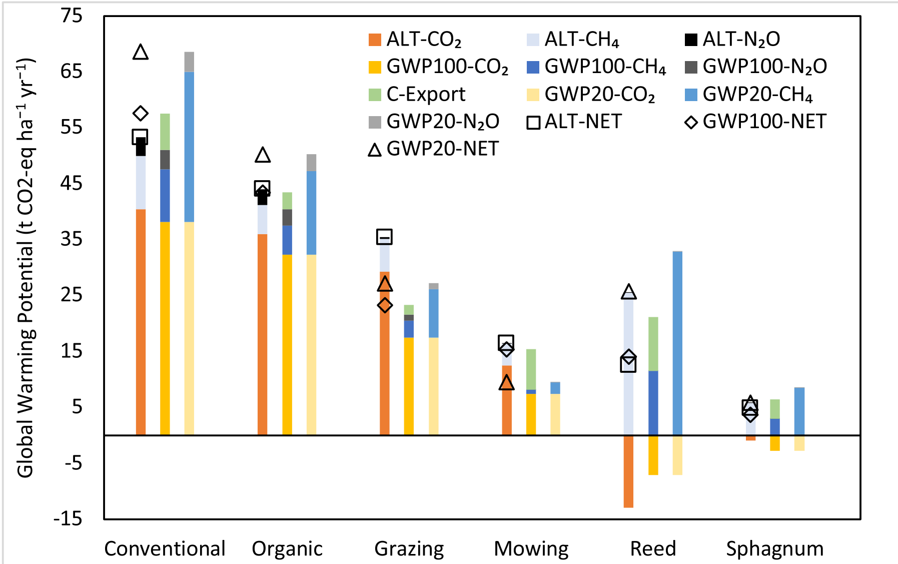


**Fig. 3.** Sensitivity analysis for model farm system inventory. 'GWP100' denotes results from the formal analysis. 'ALT' represents alternative GHG emissions accounting following the water level response functions of the statistical approach in Tiemeyer et al. (2020) using 100-year GWP. 'GWP20' is the result of applying 20-year GWP as the GHG emission metric.

regulation service when comparing to the conventional drainage-based dairy system.

At last, using 20-year GWP as an alternative GHG emission metric under a shorter time horizon brought substantially larger emphasize on $CH_4$. The subsequently higher impact from livestock emissions could lead to even larger climate regulation services when transforming from drainage-based dairy system to wet peatland systems. However, performances of the reed systems were restrained due to enlarged impact of high $CH_4$ emissions under constantly high water level. Therefore, low-intensity mowing system and Sphagnum system, at moderate water level, gave the highest climate regulation services.

### 3.2.2. Sensitivity of environmental impact assessment

Fig. 4 showed the net monetary values of model farm systems under the alternative environmental impact assessment method. Model farm systems with relative higher product values also had proportionally higher management costs. Ecosystem disservices were including GHG emissions and nitrogen pollutions led to high societal cost that consisted of major parts of the negative monetary values for almost all model farm systems except for the *Sphagnum* system. Despite lower product values, the organic dairy farm system, low-intensity grassland systems and reed system had higher net monetary values than the conventional dairy farm system due to lower management costs and ecosystem disservices. Only the reed system achieved a net balance of costs and benefit. Overall, system-level differences between model farm systems were in agreement with the ecosystem services assessment, except for the *Sphagnum* system due to the large management cost.

## 4. Discussion

This ESA evaluated the system-level differences between the environmental impact of six peatland use options based on existing data and

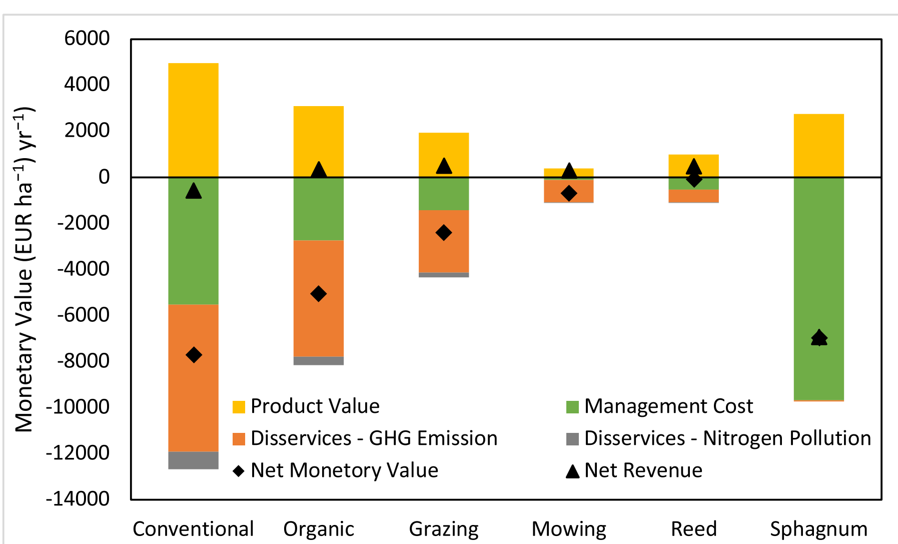


**Fig. 4.** Results of the sensitivity analysis of environmental impact assessment, including analysis of the costs of land use management and societal costs of ecosystem disservices. Net revenue is the sum of product value and management cost. Net monetary value is net revenue plus societal costs of ecosystem disservices.

knowledge of peatland soil-water-vegetation relationships. Results showed potential of co-benefits with multiple regulation and maintenance services from wetter alternative land use options following conventional drainage-based agriculture (Section 4.1). However, the ecosystem services value presented here were subject to the theoretical model system setting. Therefore, methodological uncertainties tested in the sensitivity analysis should be accounted for in specific sites or projects (Section 4.2). Nonetheless, this analysis gave insights to the understanding of the consequences of land use system transitions from conventional farming practices to wetter, more sustainable alternatives, which could support decision making in the management and planning of peatland ecosystems and landscapes (Section 4.3).

### 4.1. Potential of ecosystem services co-benefits

Ecosystem services of the model farm systems showed a clear pattern of higher overall value on higher groundwater levels, while provisioning services remained high only in the drainage-based dairy systems. Trade-offs between economic value of provisioning and other ecosystem services were frequently observed in peatland ecosystem restoration cases (e.g., Law et al., 2015). Ecological functions of peatland ecosystems rely on healthy ecohydrological dynamics (Lamers et al., 2015; Minayeva et al., 2017). Intensive dairy farming, on the other hand, requires drier soils to sustain stable grass productivity (Tiemeyer et al., 2016) and avoid damages to the pasture (Menneer et al., 2005) and animal health (Neave et al., 2022). It is clear that intensive agricultural practices that heavily rely on deep drainage and high resource inputs do not fit wetter conditions with higher groundwater levels (Tanneberger et al., 2021). The low yields from the wetter land use options were combined with low market prices resulting from under-developed markets of paludiculture biomass (Wichtmann et al., 2016), leading to provisioning services not comparable with drainage-based dairy farming. This could pose a major barrier for the transition from conventional drainage-based agriculture to sustainable alternative land uses.

However, co-benefits of multiple regulation and maintenance services in the low-intensity grassland and paludiculture systems led to higher ecosystem services value despite compromised productivity. Higher groundwater level substantially reduced field emission sources, which is in agreement with the overriding effect of water level control on GHG emissions (Evans et al., 2021). Farmyard sources of GHG emissions and nitrogen pollutions were both lower, due to the reduced or eliminated animal stocking. The soil water storage function of rewetted peatland was observed to be limited due to loss of peat thickness and state of degraded peat soils (Liu et al., 2022a). Our analysis also found limited potential of soil water storage in rewetted systems despite optimistic assumptions on the recovery of soil properties (Ahmad et al., 2020), because of the soil pore space available under high groundwater level. The overall water regulation service, however, was high because the vegetation with better flood tolerance allows surface water storage that could act as a temporary pool to mitigate flood peaks (Gao et al., 2016) without severely damage crop production. Matching of groundwater level, vegetation cover and habitat subsidy values also reflected the benefit of raised water level in restoring the biodiversity of degraded peatlands (Lamers et al., 2015). Such co-benefit of regulation and maintenance services was discovered from a number of peatland rewetting studies. For example, Renou-Wilson et al. (2019) showed that rewetting on an Irish cut-away peatland led to the return of net carbon sink function and regeneration of species typical of natural sites. Geurts et al. (2019) summarized co-benefits of emission reduction and nutrient removal from multiple paludiculture crops. Capitalizing benefits from multiple ecosystem services is essential in supporting the sustainability of peatland uses and the land use change needed.

Notably, organic dairy farming was widely studied as an effective land use option to reduce environmental impact of intensive dairy farming, providing benefits such as mitigation of emissions (Weiske et al., 2006) and improving biodiversity (Power and Stout, 2011). However, previous studies rarely differentiate between mineral and organic soils. Empirical evidence showed that $CO_2$ emission from organic dairy farms remained high due to peat oxidation under intensive drainage practices (Weideveld et al., 2021). Our analysis showed higher climate and nutrient regulation services from the organic system than the conventional system, but not comparable to the overall regulation and maintenance services from the grassland and paludiculture systems. Therefore, sustainable transition of agricultural peatlands requires more drastic changes than switching to organic practices.

### 4.2. Uncertainties in the ESA

Results of this ESA was subject to the data and assumptions integrated in the methods of the inventory analysis and ecosystem services assessment. Emission factors, indicator values, and relationships adopted from literature were appropriate under the theoretical model farm system setting. Different data and relationships suitable to different management regimes and environmental conditions could lead to uncertainties in the ESA results. This needs to be accounted for when assessing environmental impact of systems from specific sites and cases.

In the sensitivity analysis, we changed the scope of the inventory analysis by expanding the system boundary with the inclusion of carbon export, which did not change the fact that field-source $CO_2$ dominated the GHG emissions. A cradle-to-grave LCA of milk production (Thoma et al., 2013) showed that enteric methane and manure management remained as the largest contributors in the GHG budget despite emissions accounted from various sources throughout the life cycle. Therefore, sensitivity of the scale of the inventory analysis is not likely to alter the pattern of the ecosystem services.

Alternative choices of data also affected the results of the ESA. In the sensitivity analysis, change of emissions factors and GHG emission metrics drew larger emphasizes on the effects of $CH_4$ emission, while not changing the high climate regulation services provided by the rewetted systems. $CH_4$ emission is also strongly influenced by the management properties. The original analysis represented situations under good management practices. For example, the low $CH_4$ emission factors around 1–2 t $CO_2$-eq $ha^{-1}$ $yr^{-1}$ on low-intensity grassland systems were also observed on other rewetted peat soils (Karki et al., 2016). However, a poorly managed groundwater level fluctuation or prolonged flooding events on these grassland types could lead to substantially higher $CH_4$ emissions. For example, constant flooding of grassland on rewetted agricultural peatland could lead to $CH_4$ emissions up to 30 t $CO_2$-eq $ha^{-1}$ $yr^{-1}$ (Huth et al., 2021; Kandel et al., 2020). Such extreme can be prevented by certain management practices, such as topsoil removal prior to rewetting that could reduce $CH_4$ emissions by factor 30–400 (Huth et al., 2020); and sulphate addition into the soil solution that could suppress methane emission via competing microbial activities (Davidson et al., 2021; Dowrick et al., 2006). Global analysis also showed the overall GHG reduction effect of rewetting despite higher $CH_4$ emissions (Günther et al., 2020). Therefore, it is reasonable to argue that given the sensitivity of GHG emission factors, the climate regulation services will remain high in the low-intensity grassland and paludiculture systems.

Sensitivity of other indicator values of the inventory analysis could also affect the ecosystem services assessment, but not likely to alter the system-level differences between land use options. For example, nitrogen leaching estimates were conservative for drained peatland, as peat decomposition could lead to nitrogen export to surface water measured at 38 kg N $ha^{-1}$ (van Beek et al., 2007). This could only lead to even higher nutrient regulation services from alternative land use options, strengthening the current conclusions. Monetary values of ecosystem services are also sensitive to spatial and temporal variabilities. Carbon price in the European market is lower than the data we adopted from case studies at around EUR 50 on average for the year 2021 (Sechi et al., 2022). However, it was increased to EUR 100 by August 2022 (www.statista.com), and is expected to continue increasing significantly in the coming decade under the net zero target, potentially leading to higher monetary values of the climate regulation services. With increasing economic losses from flooding observed under the changing climate (Kundzewicz et al., 2014), one can expect that the value of water storage as flood buffer would also increase. On the other hand, more frequent drought event due to climate change (Hari et al., 2020) will also increase the value of groundwater recharge function from the subsoil water storage of peatlands. Similarly, the management intervention required and the costs for habitat protection will differ regionally depending on the site conditions and land use history (Lamers et al., 2015), thus influence the estimation of habitat values following our methodology. Overall, it is likely that the differences of regulation and maintenance services value between drainage-based dairy farming systems and wetter alternative systems would be strengthened rather than overturned under the above-mentioned sensitivity of indicator values.

The sensitivity analysis demonstrated the uncertainties in the environmental impact assessment by applying a different approach. Assessing GHG emissions and nitrogen pollution as ecosystem disservices led to net negative values of all model farm systems, especially for the drainage-based dairy systems. This approach of assessing negative environmental impact supported the benefits of ecosystem services provided by the rewetted system. More ecosystem disservices related to traditional drainage exists. For example, land subsidence – largely caused by intensive peatland drainage – is predicted to cause an accumulated damage cost of over EUR 5 billion for infrastructure alone in the Netherlands till 2050 (Stouthamer et al., 2020); the Dutch nitrogen crisis has inflicted high costs by slowing down building industries, causing even higher economic damages (Stokstad, 2019). This way of assessing negative environmental impact was in agreement with the ecosystem services assessment that showed the benefits of rewetted system. This notion was further supported by the assessment of management costs, as the higher agricultural intensity brought higher management costs, making the conventional dairy farm system one of the least cost-effective. The *Sphagnum* system also induced large management cost, which echoed the need of financial and policy support for the transition towards this land use.

### 4.3. Incentivizing system transition

The presented ESA raised the most prominent issues for the sustainable land use transition on peatlands: the low economic feasibility due to low biomass provisioning, and the difficulty in capitalizing regulation and maintenance services. Nonetheless, possible incentives for the transition exists when considering more socio-economic factors at scales beyond the system level. First, the conventional dairy farming system is economically vulnerable, not only due to the high management costs, but also because it is heavily dependent on subsidies under the European Common Agriculture Policy (CAP) (Poczta et al., 2020; Sechi et al., 2022). This made the current business model vulnerable to technical, climatic and economic risks despite a high level of productivity (Bouttes et al., 2019). For example, perennial ryegrass as a common pasture crop is more likely to suffer from severe drop of yield under extreme weather conditions such as the drought event of 2018 in Europe (Fu et al., 2020; Liu et al., 2022b). On the other hand, low management input requirements and better resilience of the vegetation on the low-intensity grassland and paludiculture systems could perform more stably under future climate change. Although the current assessment of management cost of the *Sphagnum* system is substantially higher than the revenue, longer farm operation could potentially lead to higher yield at 405 $m^3$ $ha^{-1}$ $yr^{-1}$ (Vroom et al., 2020), which could lead to a positive net revenue at EUR 442 $ha^{-1}$. The high management cost was mainly characterized by the high establishment cost averaged and added to the annual costs (Wichmann et al., 2020), which is also likely to be reduced with developments of the farming techniques.

Meanwhile, the co-benefits of regulation and maintenance services from the wetter alternative systems hinted at potential economic incentives for the transition of systems via payments for ecosystem services (Ziegler et al., 2021). Carbon farming, aiming at improving the rate of $CO_2$ removal from atmosphere and conversion into plant biomass and soil organic matter, presents opportunities to combine paludiculture crop production with carbon crediting (Tanneberger et al., 2021). The potential use of biomass as bioenergy feedstocks can also provide substitution effect that prevents the use of fossil-based energy and materials of around 5–10 t $CO_2$-eq $ha^{-1}$ $yr^{-1}$ (Table S6). Besides the reduction of nitrogen pollution, wet crops such as reed (Geurts et al., 2020) and *Sphagnum* (Vroom et al., 2020) are also found to have nutrient removal function that could further increase the nutrient regulation services. Benefits from this multifunctionality can be captured and monetized as part of a peatland carbon scheme as well (Bonn et al., 2014). For example, MoorFuture® presented methodologies to combine carbon crediting and accounting of ecosystem services (Joosten et al., 2015).

In summary, under the pressure of the changing climate on the development of sustainable agriculture (Agovino et al., 2019), it is likely that the economic benefit of milk production and the policy support of the dairy-farming normality will be out-weighted by the ecosystem services and disservices of land use options under wetter conditions. Relocating intensive agriculture away from peatlands to mineral soils is likely to be an option, as the efficiency of agriculture production is higher on the drier and more fertile mineral soils (Sechi et al., 2022). This could also avoid the leakage effect that displace the ecosystem disservices of intensive agriculture on other non-accounted ecologically sensitive areas (Ewers and Rodrigues, 2008). However, without accounting for the societal costs of environmental damages in the current agricultural business model and capitalizing the ecosystem services values as economic benefits, it is not likely that the farmers will change their current production systems.

## 5. Conclusions

This study identified the differences in ecosystem services of land use options under a gradient of water levels. Our ESA methods provided a simple approach that integrated existing knowledge to shed light on the system-level differences between typical land use options on peatlands. In conclusion, the continuation of current way of dairy farming on peatland under conventional drainage is not likely to achieve major sustainability improvements. The low ecosystem services value, large costs and disservices severely offset the value of dairy production. Conversion to organic farming provided notable climate and nutrient regulation services. However, more drastic land use changes were needed to reverse the damage of traditional drainage. Meanwhile, the paludiculture farm systems that transformed from animal keeping to biomass utilization provided far larger regulation and maintenance services comparing to the drainage-based farming systems, although the level of biomass provisioning was not comparable to dairy farming, let alone the high establishment and management costs. Fundamental changes in land and water management are required for the sustainable use of peatlands, with critical need of financial and policy support for the land use conversion. Capitalization of both societal costs of ecosystem disservices from intensive agriculture and ecosystem services benefits from alternative land use options are required.

## CRediT authorship contribution statement

**Weier Liu:** Conceptualization, Methodology, Writing – original draft. **Christian Fritz:** Writing – review & editing, Methodology, Validation, Supervision. **Jasper van Belle:** Writing – review & editing, Methodology, Validation. **Sanderine Nonhebel:** Writing – review & editing, Supervision.

## Data availability

Data collected and used in this analysis is presented in the supplement material.

## Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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## Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.scitotenv.2023.162534.

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